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Blunt-nosed Leopard Lizard, *Gambelia sila*
Drawing by Julie Brown, California Department of Fish and Game, 1989

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VARIATION IN USE OF THE KLAMATH RIVER ESTUARY BY JUVENILE CHINOOK SALMON

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Use of the Klamath River estuary by young-of-the-year (YOY) chinook salmon, *Oncorhynchus tshawytscha*, varied between high (1993) and low (1994) river flow years. From May to September, mean flows in the Klamath River were 400 m^3/s in 1993 and 109 m^3/s in 1994. In the lower Klamath River estuary, YOY chinook salmon catch-per-unit-effort was higher during 1994 than during 1993. Also, weekly mean fork lengths of YOY chinook salmon in the Klamath River estuary after mid-July were significantly smaller in 1994 than 1993. These observations suggest that more estuarine rearing by YOY chinook salmon took place in the low flow year of 1994 than the high flow year of 1993, potentially because of better up-river rearing conditions in 1993. In 1993, most YOY chinook salmon reached the estuary at a size large enough to immediately enter the ocean. In 1994, many juvenile chinook salmon may have been forced downstream by increasing water temperatures before they reached optimal size for ocean entry. Thus, they were more likely to rear in the estuary.

INTRODUCTION

Chinook salmon, *Oncorhynchus tshawytscha*, life history patterns are the most complex of the Pacific salmon. In California and Oregon, life history varies from rearing in freshwater for a short period before migrating to the ocean (ocean type) to rearing in the stream for a full year before seaward migration (stream type); intermediate life history patterns also exist. Freshwater life history of juvenile chinook salmon is suspected to strongly influence survival and return as adults (Nicholas and Hankin 1989, Healey 1991). Growth rate, emigration timing, distribution, and rearing patterns determine when and at what size juveniles enter the marine environment. Examination of these factors will help to identify important habitats and critical times to better understand chinook salmon population dynamics.

Though many estuaries along the Pacific Coast of North America are important rearing areas for some salmonid species (Reimers[1] 1971, Healey 1980, Kjelson et al.

[1] Reimers, P.E. 1971. The length of residence of juvenile fall chinook in Sixes River, Oregon. Ph.D. Dissertation, Oregon State University, Corvallis, Oregon, USA.

1982, Levy and Northcote 1982, Myers and Horton 1982), evidence about the extent to which chinook salmon rear in the Klamath River estuary is contradictory. Evidence that some emigrating chinook salmon rear for a period of time in the Klamath River estuary is provided by mark-recapture studies (CDFG[2] 1993a, CDFG[3] 1994a), scale circuli analysis (Snyder 1931), recapture of coded-wire-tagged chinook salmon in the estuary up to 4 months after their release into Hunter Creek within a few kilometers of the estuary (Wallace 1995), and peak chinook salmon catches in the lower estuary typically occurring 1–2 weeks later than in the upper estuary (CDFG[4] 1992a). Conversely, other studies have concluded that extended rearing of young-of-the-year (YOY) chinook salmon rarely takes place in the Klamath estuary. Krakker[5] (1991) found that YOY chinook salmon captured from the lower estuary and the lower mainstem river upstream of the estuary were similar in size and had similar emigration timing, thus concluding that few fish reared in the estuary. Sullivan[6] (1989) used scale circuli analysis to describe juvenile life histories of fall chinook salmon from the Klamath basin and concluded that they did not rear extensively in the estuary. These contradictory conclusions about the use of the Klamath estuary by juvenile chinook salmon may result from the fishes' response to annual changes in physical or biological processes, thereby resulting in variable patterns of estuarine use by chinook salmon.

Pacific Coast estuary conditions exhibit wide annual variations (Simenstad and Wissmar 1984) that probably affect chinook salmon rearing patterns in estuaries. For example, Healey (1980) found annual differences in abundance and size of chinook salmon fry in the Nanaimo River estuary, British Columbia. Other researchers have also noted annual differences in distribution or size of juvenile chinook salmon captured in estuaries (Kjelson et al. 1982, Simenstad and Wissmar 1984, McCabe et al. 1986).

Physical conditions in the relatively small Klamath River estuary are dominated by river flow and vary greatly both seasonally and annually (CDFG[7] 1992b,

[2] California Department of Fish and Game. 1993a. Utilization of the Klamath River estuary by juvenile salmonids. Annual Performance Report. Federal Aid in Sport Fish Restoration Act. Project No. F-51-R-5, Subproject No. IX, Study No. 10, Job No. 3.

[3] California Department of Fish and Game. 1994a. Length of residency of juvenile chinook salmon in the Klamath River estuary. Annual Performance Report. Federal Aid in Sport Fish Restoration Act. Project Number F-51-R-6, Project No. 32, Job No. 4.

[4] California Department of Fish and Game. 1992a. Utilization of the Klamath River estuary by juvenile salmonids. Annual Performance Report. Federal Aid in Sport Fish Restoration Act. Project No. F-51-R-4, Subproject No. IX, Study No. 10, Job No. 3.

[5] Krakker, J.J., Jr. 1991. Utilization of the Klamath River estuary by juvenile chinook salmon (*Oncorhynchus tshawytscha*), 1986. M.S. Thesis, Humboldt State University, Arcata, California, USA.

[6] Sullivan, C.M. 1989. Juvenile life history and age composition of mature fall chinook salmon returning to the Klamath River, 1984-1986. M.S. Thesis, Humboldt State University, Arcata, California, USA.

[7] California Department of Fish and Game. 1992b. Assessment of fish habitat types within the Klamath River estuary. Annual Performance Report. Federal Aid in Sport Fish Restoration Act. Project Number F-51-R-4, Subproject No. IX, Study No. 22, Job No. 1.

CDFG[8] 1993b, CDFG[9] 1994b, CDFG[10] 1995). The magnitude and timing of river flow probably determines the amount of rearing habitat available to juvenile chinook salmon in the basin, which, in turn, influences emigration timing and fish condition upon entering the estuary.

Studies of the Klamath River estuary since 1986 have documented emigration timing, size, distribution, habitat use, and diets of juvenile salmonids (Wallace[11] 1993, Wallace 1995, Wallace and Collins[12] 1995a, Wallace and Collins[13] 1995b). Intensive sampling in the Klamath River estuary in 1993 and 1994 to determine juvenile chinook salmon length of residence provided an opportunity to compare patterns of estuary use during high (1993) and low (1994) river flow years. This study reports our observations about emigration timing and patterns, relative abundance, and size of YOY chinook salmon in the Klamath River estuary in those 2 yr. Based on our observations of the behavior of juvenile chinook salmon at greatly different river flows, we provide insight on how YOY chinook salmon use the estuary and speculate how they reacted to flow conditions throughout the rest of the basin. This may help explain why different studies have come to different conclusions about the importance of the Klamath River estuary as a rearing area for YOY chinook salmon.

METHODS

Study Area

The Klamath River enters the Pacific Ocean about 51 km south of the California-Oregon border. Its estuary is relatively short and small when compared to the size of the watershed. The estuary provides numerous habitat types, even

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[8] California Department of Fish and Game. 1993b. Assessment of fish habitat types within the Klamath River estuary. Annual Performance Report. Federal Aid in Sport Fish Restoration Act. Project Number F-51-R-5, Subproject No. IX, Study No. 22, Job No. 1.

[9] California Department of Fish and Game. 1994b. Seasonal water quality monitoring in the Klamath River estuary. Annual Performance Report. Federal Aid in Sport Fish Restoration Act. Project No. F-51-R-6, Category: Surveys and Inventories, Project No. 33, Job No. 2.

[10] California Department of Fish and Game. 1995. Seasonal water quality monitoring in the Klamath River estuary. Annual Performance Report. Federal Aid in Sport Fish Restoration Act. Project No. F-51-R-6, Category: Surveys and Inventories, Project No. 33, Job No. 2.

[11] Wallace, M. 1993. Distribution, abundance, size, and coded-wire tag recovery of juvenile chinook salmon in the Klamath River estuary, 1986-1989. Final Performance Report. Federal Aid in Sport Fish Restoration Act. Project No. F-51-R; Subproject IX: Study No. 10; Job No. 3.

[12] Wallace, M. and B.W. Collins. 1995a. Food habits and preferences of juvenile chinook salmon in the Klamath River estuary. Final Performance Report. Federal Aid in Sport Fish Restoration Act. Project No. F-51-R; Project No. 32; Job No. 6.

[13] Wallace, M. and B.W. Collins. 1995b. Habitat type utilization of juvenile salmonids in the Klamath River estuary. Final Performance Report. Federal Aid in Sport Fish Restoration Act. Project No. F-51-R; Project No. 32; Job No. 7.

though it lacks the extensive tide flats and tidal marshes found in most larger estuaries (CDFG[7] 1992b, CDFG[8] 1993b). Saltwater intrusion varies seasonally and is controlled by freshwater flow and a sand berm which forms in late summer at the river mouth.

For this study, we divided the Klamath River estuary into lower and upper sections. The lower estuary (river km [rkm] 0–2.4) experiences tidal fluctuation up to 2 m and brackish water (15–30‰) is usually present along the bottom from May through October (CDFG[7] 1992b, CDFG[8] 1993b, CDFG[9] 1994b, CDFG[10] 1995). In the upper estuary (rkm 2.4–6.4), brackish water usually extends upstream to about rkm 4.8 at high tide, but reaches as far as rkm 6.4 when high tides coincide with low river discharge (CDFG[8] 1993b). However, a layer of fresh water 1–2 m deep is found along the surface throughout most of the estuary, causing shallow littoral areas to be primarily freshwater habitat.

Sampling

Sampling locations and methods were the same in both years of our study. We attempted to sample the upper and lower estuary each week. We established four transects in the upper estuary that consisted primarily of sand, gravel, and cobble flats and heavily vegetated cut banks (Fig. 1). We sampled each transect for 10 min at night using a boat-mounted electrofisher. The electrofisher was powered by a 5.0-kilowatt generator. The anodes were two 0.9-m diameter circular clusters of six

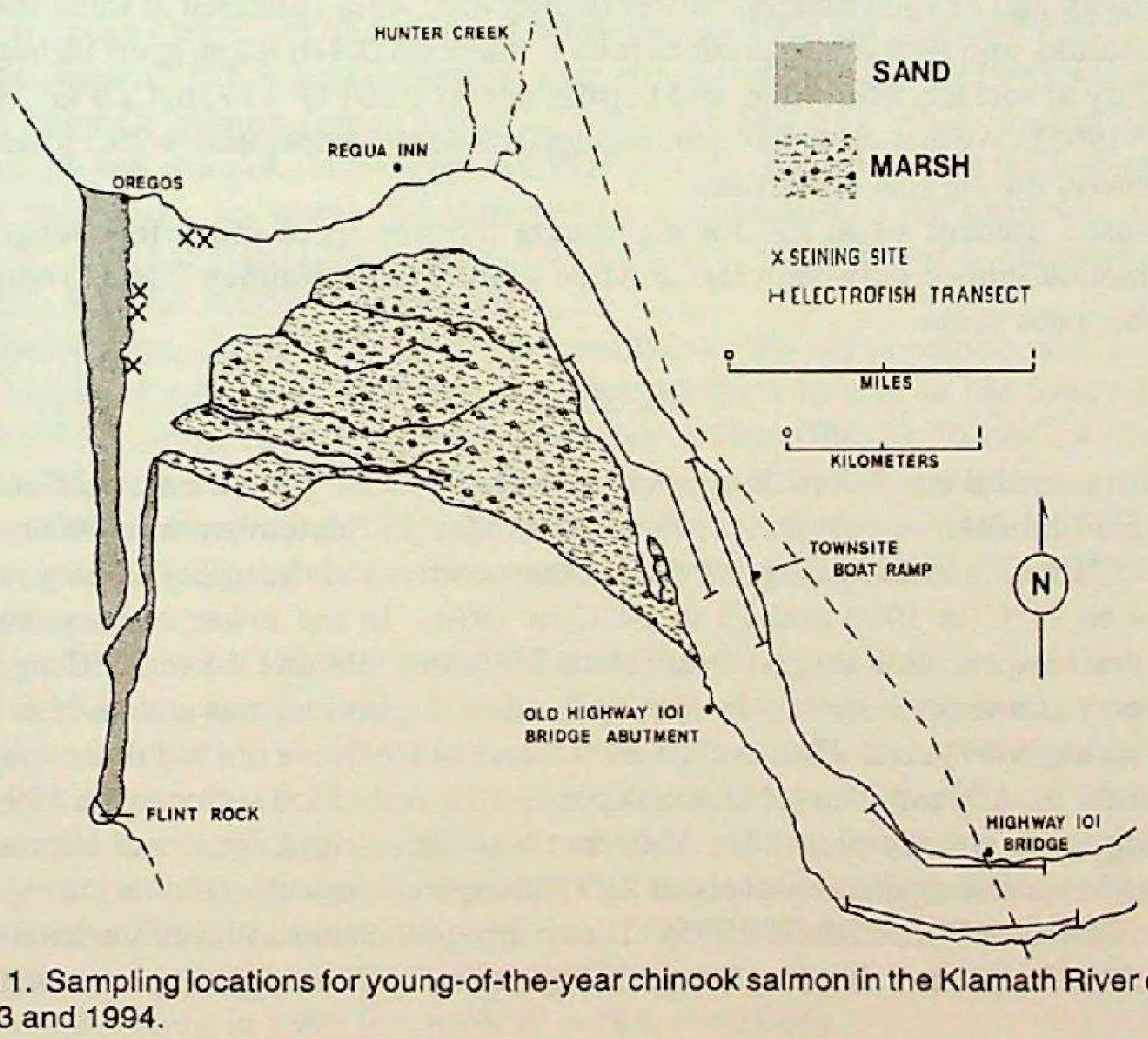

Figure 1. Sampling locations for young-of-the-year chinook salmon in the Klamath River estuary in 1993 and 1994.

6.4-mm diameter stainless steel cables that were extended by booms to 2.4 m in front of the boat. The cathode was an array of seventeen 3.2-mm diameter stainless steel cables hung 152 mm apart and attached to the bow of the boat. We sampled fish at 250–300 v (passing 3–5 amps) at 120 pulses/s DC in 5- to 7-s bursts. Upstream and downstream boundaries were established for each transect to minimize the variation in the amount of area sampled. Catch-per-unit-effort (CPUE) was calculated as the number of fish captured per minute shocked. We electrofished in the upper estuary because it allowed sampling of a variety of habitats and was more efficient than beach seining at capturing larger juvenile salmonids.

In the lower estuary, during daytime, we deployed a 45.7-m x 3.1-m beach seine with 6.4-mm mesh from the bow of a 4.9-m boat. We could not electrofish the lower estuary due to widespread presence of salt water. We sampled five standard locations consisting of sand and gravel flats and sand beaches (Fig. 1). The length and width of each haul was estimated to calculate the area seined. Catch-per-unit-effort was calculated as the number of fish captured per 100 m^2 seined.

Captured salmonids were anesthetized with quinaldine sulfate prior to measurements. Fork lengths were recorded to the nearest millimeter for up to 30 fish per species per transect or haul. All salmonids were counted and examined for marks.

River flows were determined from a California Department of Water Resources gaging station located near the mouth of Turwar Creek approximately 1.6 km above the estuary. We collected monthly water quality data at high and low tides using conductivity and oxygen meters. Water quality data were collected at three stations (right, middle, and left) along predetermined transects 0.4–0.8 km apart throughout the estuary at surface, midwater, and bottom depths (CDFG[8] 1993b, CDFG[9] 1994b, CDFG[10] 1995). We also routinely sampled surface water temperature with a hand-held thermometer during fish collections.

We used Student's-t to test for significant differences in mean fork lengths of YOY chinook salmon between years and two-tailed Mann-Whitney U test to compare CPUE between years.

RESULTS

Environmental conditions in the Klamath River estuary were much different in 1993 than in 1994. Average river flow from May to September was 400 m^3/s in 1993 and 109 m^3/s in 1994 (Fig. 2). Water temperatures in the upper estuary ranged from 14 to 21°C in 1993 and 15 to 24°C in 1994. In the lower estuary, surface freshwater temperatures ranged from 14 to 21°C in 1993 and 16 to 23°C in 1994. However, water temperature in the salt wedge near the bottom was normally 5–10°C cooler during both years. Due to high river flows in 1993, we did not detect any salt water in the estuary until July, but it was present the rest of the summer. In 1994, the salt wedge was already present in May, but was absent in August and September, resulting in water temperatures of about 21°C throughout the water column during those months (CDFG[9] 1994b, CDFG[10] 1995). The change in location and configuration of the river mouth likely kept salt water from entering the lower estuary during late summer in 1994.

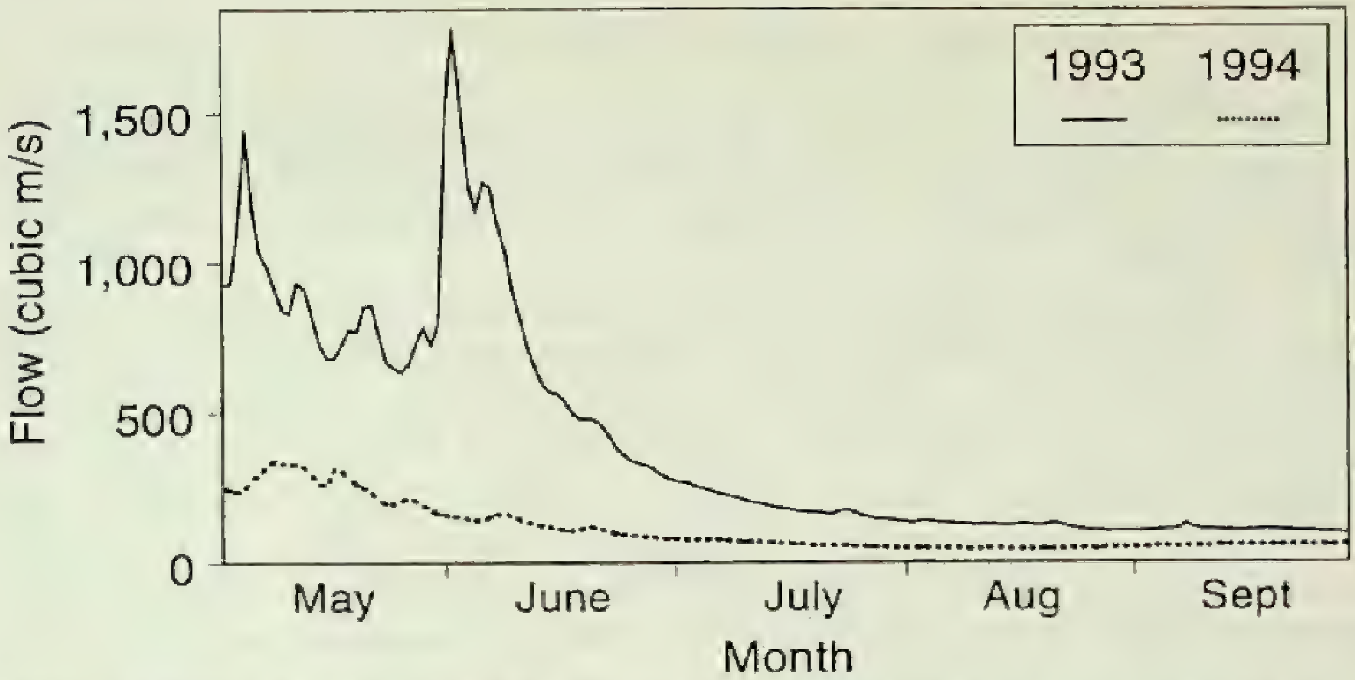

Figure 2. Average daily Klamath River flows at the Turwar gaging station from May to September 1993 and 1994.

The average CPUE of YOY chinook salmon in the upper estuary during 1993 was significantly higher than in 1994 (U = 237, df = 32, P <0.001). Although CPUE did not differ between years in the lower estuary (U = 215, df = 36, P >0.20), it was significantly higher from mid-July to early September in 1994 than in 1993 (U = 54.5, df = 14, P <0.05). In the upper estuary, during 1993, we captured 3,411 chinook salmon (mean CPUE = 6.61 fish/min), while during 1994 we captured 1,265 chinook salmon (mean CPUE = 1.54 fish/min). In the lower estuary, during 1993, we captured 1,194 chinook salmon (mean CPUE = 3.38 fish/100 m^2), while during 1994 we captured 2,468 chinook salmon (mean CPUE = 4.86 fish/100 m^2). In the lower estuary from mid-July to early September, we captured 190 chinook salmon (CPUE = 1.11 fish/100 m^2) in 1993 and 1,596 chinook salmon (mean CPUE = 9.27 fish/100 m^2) in 1994.

The average CPUE of YOY chinook salmon in the upper estuary also remained high longer in 1993 than in 1994, but the opposite was true in the lower estuary, where CPUE remained high longer in 1994 than in 1993 (Fig. 3). In 1993, we obtained 50% of our upper estuary YOY chinook salmon catch about a month later than 50% of our lower estuary catch (Fig. 4). In contrast, in 1994, we obtained 50% of our upper estuary YOY chinook salmon catch about a month earlier than 50% of our lower estuary catch.

Juvenile chinook salmon in the Klamath River estuary differed in size between years only in late summer (Fig. 5). Mean lengths of juvenile chinook salmon from June to mid-July were similar in both years: 87.3 mm in 1993 vs. 87.6 mm in 1994 in the upper estuary (t = 0.31, df = 810, P >0.20) and 85.4 mm in 1993 vs. 86.0 mm in 1994 in the lower estuary (t = 0.94, df = 846, P >0.20). However, after mid-July, mean length in the upper estuary in 1993 (95.8 mm) was significantly larger (t = 14.51, df = 965, P <0.001) than in 1994 (85.5 mm). The same was true in the lower estuary, where the mean length of 100.2 mm in 1993 was significantly larger than the mean length of 88.5 mm in 1994 (t = 9.14, df = 713, P <0.001).

DISCUSSION

The differences in timing and magnitude of our YOY chinook salmon catches in the upper and lower estuary between years may be explained by either annual differences in emigration timing or differences in number of chinook salmon rearing in the two areas.

A change in emigration timing rather than increased numbers of rearing chinook salmon was the most probable cause for higher catches observed in the upper estuary in 1993. The U.S. Fish and Wildlife Service (USFWS) also noted later emigration of chinook salmon at traps on the mainstem Klamath River in 1993 than in 1994 (J. Lang, USFWS, pers. comm.). Higher river flows in 1993 likely created more rearing habitat (Bjornn and Reiser 1991) and cold-water refugia upstream of the estuary and, therefore, may have allowed more juvenile chinook salmon to rear later into the year before emigrating. Flows from cold-water tributaries such as the Salmon River, Scott River, and Indian Creek were higher in 1993 than in 1994 (Palmer et al.

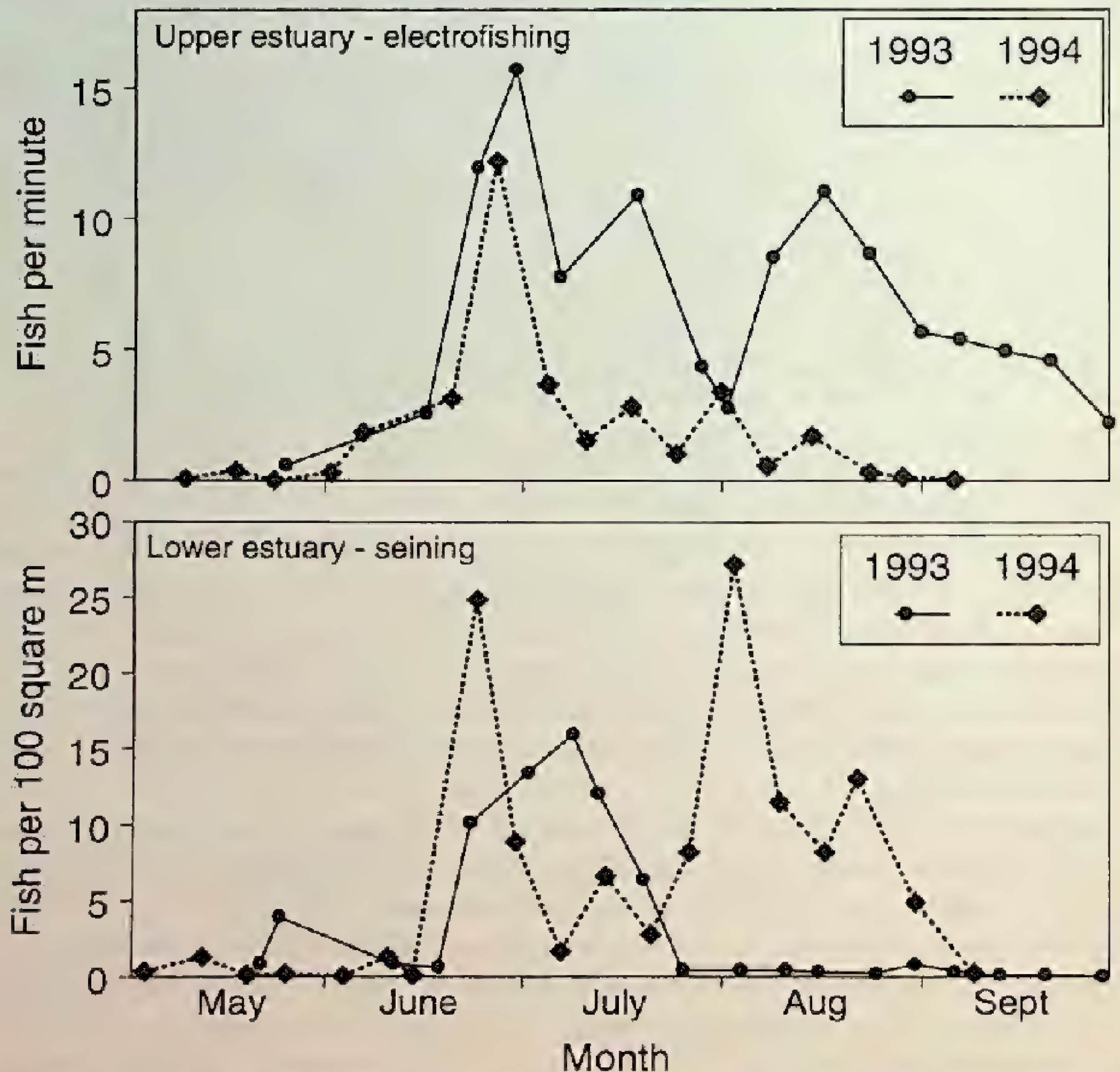

Figure 3. Catch-per-unit-effort of young-of-the-year chinook salmon from the upper and lower Klamath River estuary in 1993 and 1994.

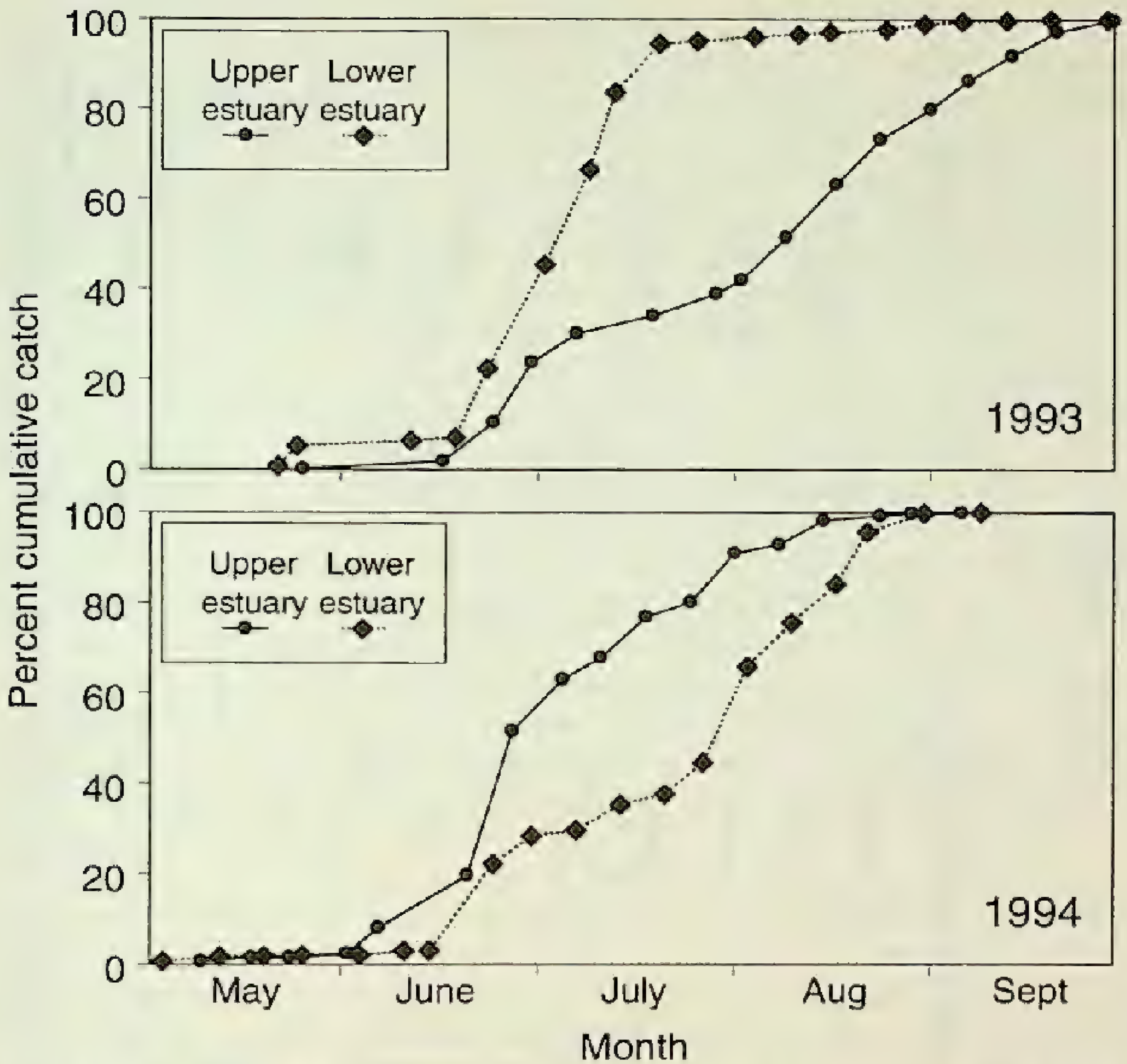

Figure 4. Cumulative catch of young-of-the-year chinook salmon from the upper and lower Klamath River estuary in 1993 and 1994.

1993, Ayers and others 1994), which likely increased the area of cool-water habitat available to juvenile chinook salmon. High concentrations of juvenile chinook salmon were observed adjacent to the mouths of cold-water tributaries of the Klamath and Trinity rivers in the mid-1980s (T. Mills, California Department of Fish and Game, pers. comm.). These areas may act as cool-water refugia from warm mainstem water and provide important rearing areas for juvenile chinook salmon. Finally, the significantly larger mean length in 1993 is consistent with more favorable mainstem river conditions in 1993 than in 1994. However, whether the larger size was due to more food in 1993, less cool-water refugia area inhibiting chinook salmon growth in 1994, or some other reason is not known.

Several factors suggest that higher upper-estuary catches in 1993 were not the result of an increase in the number of YOY chinook salmon rearing there. We measured water temperatures up to 21°C during monthly water quality sampling and up to 22.5°C during fish sampling, which, though not as high as in 1994, still approach

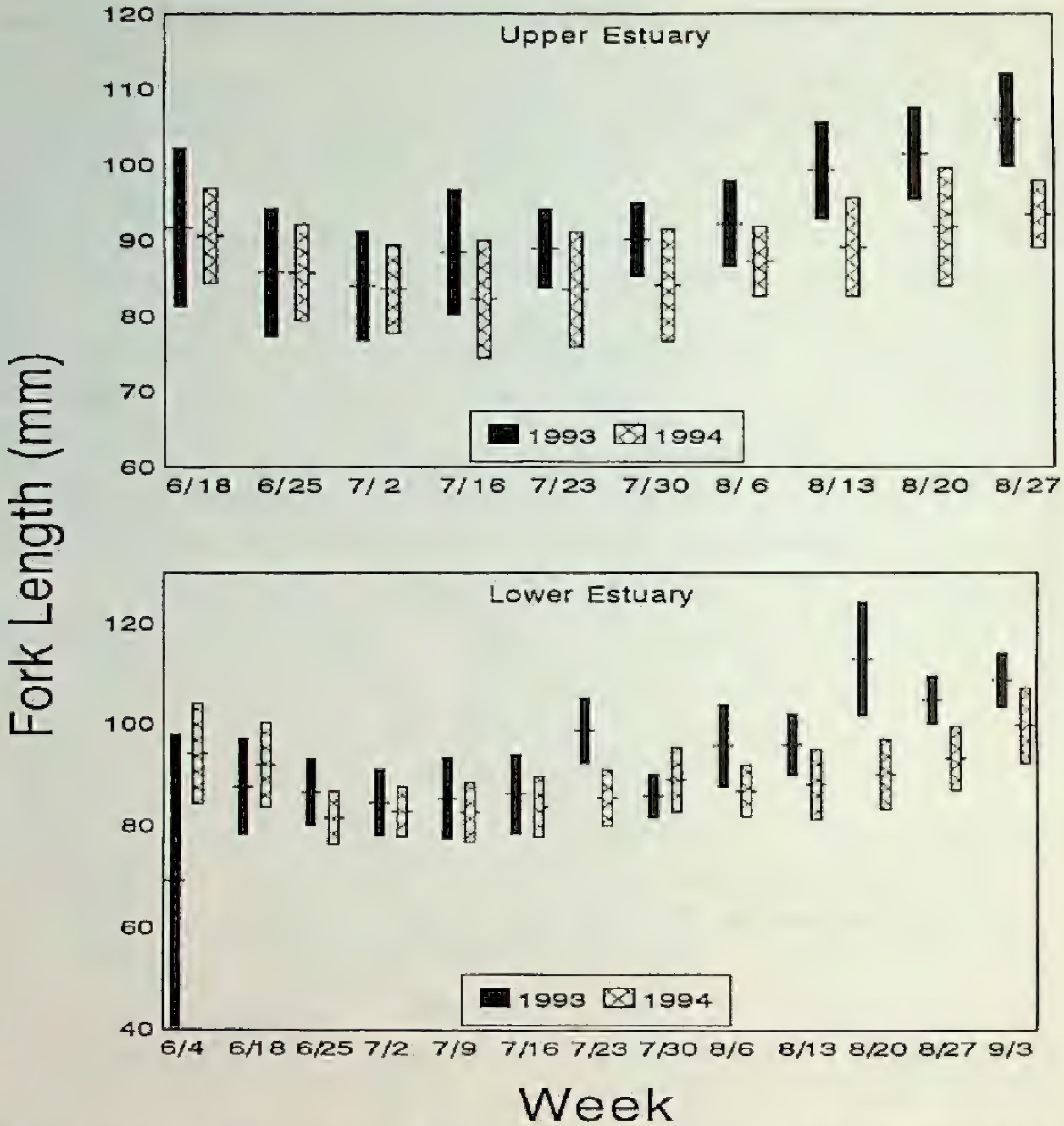

Figure 5. Weekly mean fork length ± 1 standard deviation of young-of-the-year chinook salmon captured from the upper and lower Klamath River estuary in 1993 and 1994.

the upper lethal limit for juvenile chinook salmon (Bjornn and Reiser 1991). Also, it is likely that catches of YOY chinook salmon in the lower estuary would have increased as the greater number of rearing fish dispersed throughout the entire estuary. Finally, the large size of fish captured in the upper estuary in 1993 suggested that they were already large enough for ocean entry by the time they reached the estuary (Nicholas and Hankin 1989) and, therefore, less likely to rear there. We were probably more likely to intercept these larger fish while electrofishing in the upper estuary than seining the lower estuary because they are more vulnerable to electrofishing than seining (Reynolds 1996) and less likely to occupy nearshore areas sampled by our seines (Myers and Horton 1982).

Our observations suggest that, while juvenile chinook salmon spent little time rearing in the lower Klamath River estuary in 1993, they reared there more extensively

in 1994. In 1993, relatively large numbers of chinook salmon continued to be caught in the upper estuary after lower estuary catches dropped to low levels, suggesting that most fish were moving quickly through the lower estuary or, possibly, residing in offshore areas of the estuary, thereby avoiding our seines. However, we observed different catch patterns in 1994. We captured relatively low numbers of chinook salmon in the upper estuary after mid-July, while high catches continued in the lower estuary. This suggests that YOY chinook salmon passed through the upper estuary earlier and spent a longer period of time in the lower estuary in 1994 than in 1993. Also, in 1994, cumulative and peak catches of chinook salmon occurred about a month later in the lower compared to upper estuary. If this later peak in the lower estuary was due to increased emigration, rather than increased rearing, we would have seen a similar increase in our upper estuary chinook salmon CPUE.

Increased estuary rearing by chinook salmon in 1994 was likely due to unfavorable rearing conditions upstream; thus, YOY chinook salmon probably reached the estuary at a smaller size than in 1993. After mid-July 1994, YOY chinook salmon were significantly smaller than during the same time in 1993. Low river flows in 1994 may have created poor rearing conditions that reduced chinook salmon growth rate or forced earlier emigration at a smaller than optimum size for ocean entry. Juvenile chinook salmon from Oregon coastal rivers that enter the ocean in late summer and early fall at 120–160 mm experience the highest ocean survival, though typically the greatest number of chinook salmon enter the ocean at 90–100 mm in length (Nicholas and Hankin 1989) .

Assessing the prevalence of estuarine rearing by comparing chinook salmon lengths between the upper and lower estuary alone does not appear to be a reliable method. In both 1993 and 1994, sizes were similar between the upper and lower estuary. One reason is that size differences may be partially masked by our gear selecting smaller fish in the lower estuary. Wallace[11] (1993) reported that, when sampling the same areas of the Klamath River estuary, chinook salmon captured by electrofishing typically averaged 2–3 mm larger than those captured by beach seines. Another reason may be that the smaller chinook are more likely to rear in the estuary than larger chinook in order to attain adequate size for ocean entry. Therefore, we may be comparing the size of all chinook entering the upper estuary with predominately smaller, rearing chinook in the lower estuary. When fish reach a suitable size for ocean entry, it is likely that they migrate out of the lower estuary to the ocean.

Other explanations for the smaller size of chinook salmon observed in 1994 than in 1993 are that higher abundance in the lower estuary depleted the available food supply and that a change in the location and configuration of the river mouth kept cool saltwater from entering the lower estuary later in summer 1994 and inhibited chinook salmon growth as water temperatures reached 20–23°C (CDFG[7] 1992b, CDFG[8] 1993b, CDFG[9] 1994b, CDFG[10] 1995). The former hypothesis is supported by evidence that, in 1991 and 1992, abundance of preferred chinook salmon prey items was lowest in the summer immediately after peak catches of chinook salmon (Wallace and Collins[12] 1995a).

Substantial annual variations in physical and biological processes occur in Pacific Coast estuaries (Simenstad and Wissmar 1984) and this may explain why some studies have shown significant estuarine rearing by YOY chinook salmon (Reimers[1] 1971, Healey 1980, Kjelson et al. 1982, Levy and Northcote 1982, Myers and Horton 1982), whereas others have concluded that relatively little estuarine rearing takes place (Schluchter and Lichatowich 1977, Sullivan[6] 1989, Krakker[5] 1991). All of these studies were conducted over relatively few years and may not have encompassed the full range of variability in these processes. Although our study is no exception, it does suggests that changes in the Klamath River estuary, including annual differences in river flow and brackish water conditions of the lower estuary, elicited changes in estuarine use by YOY chinook salmon. Therefore, conclusions about the importance of estuaries to chinook salmon production need to be based on a longer time series that includes a wide range of variability in physical and biological processes. This may be especially true in smaller estuaries like the Klamath, where annual physical conditions can vary greatly.

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UTILITY OF 10-DAY CENSUSES TO ESTIMATE POPULATION SIZE OF BLUNT-NOSED LEOPARD LIZARDS

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The blunt-nosed leopard lizard, *Gambelia sila*, is an endangered species that increasingly is losing habitat within its range in the San Joaquin Valley of California. Determining local population size of this species sometimes is necessary for its conservation. Measuring population size from short-duration censuses can save time and money compared to total counts or mark-recapture methods, but the census must be shown to be accurate. From 1990 to 1994, we completed full-season censuses of blunt-nosed leopard lizards that were marked and recaptured on two plots on the Elkhorn Plain, California. We compared estimates of population size from full-season censuses (considered a true estimate of abundance) to 10-day counts of the adult/yearling cohort (active April–July) and hatchling cohort (active July–October). We found that 10-day censuses can be used to accurately index population size of blunt-nosed leopard lizards, at least in foothill habitat.

INTRODUCTION

Monitoring fluctuations in population size of reptiles is increasingly becoming an important part of species management, particularly for species that are in jeopardy of extinction. Estimating the size of populations at various sites is often necessary to monitor effects of environmental variation as well as to determine effects due to management practices, such as grazing by livestock. Several methods have been used to estimate abundance of lizard populations, including total counts, estimates based on mark-recapture, and time-constrained searching (Dunham et al. 1988, Campbell and Christman 1982, Corn and Bury[1] 1990).

[1] Corn, P.S., and R.B. Bury. 1990. Sampling methods for terrestrial amphibians and reptiles. U.S. Forest Service, General Technical Report PNW-GTR-256, Portland, Oregon, USA.

Mark-recapture of individuals is perhaps the most accurate way of estimating population size and also allows collection of demographic and morphologic data because animals have to be caught. However, the requirement to catch individuals over an extended period of time limits the value of this method for monitoring when compared to a quicker method that does not require the handling of individuals.

Simply counting lizards over a period of several days without catching or marking individuals would be a quicker and less expensive method of indexing abundance, but would only be useful if it accurately reflected population size (Turner 1977). A simple counting method, termed the cruise method, was used to index population size of three species of diurnal lizards at six sites in Big Bend National Park, Texas (Degenhardt 1966), but its accuracy was not tested (Turner 1977). The cruise method involves walking a staked plot for a certain number of days and simply counting all the lizards observed. This count is presumed to reflect true abundance.

A modified cruise method has been used for the past 2 decades (at the urging of management agencies) to measure abundance of the blunt-nosed leopard lizard, *Gambelia sila*, an endangered species in the San Joaquin Valley, California. The blunt-nosed leopard lizard is a large, conspicuous reptile of shrublands and sparse annual grasslands for which several population studies have been conducted in the past 3 decades (Montanucci 1965; Tollestrup 1982, 1983; Williams[2] et al. 1993; Germano et al. 1994). Because of its endangered status, there is a need to determine presence and sometimes measure abundance of blunt-nosed leopard lizards when development is proposed within the range of this species. Estimating abundance of this lizard also is important to agencies and conservation groups that manage preserves set aside to protect this species.

The standard technique used to index density of blunt-nosed leopard lizards is a 10-day census developed by Tollestrup[3] (1976), which is based on the cruise method described by Degenhardt (1966). Tollestrup modified the cruise method by using a larger plot size (8.1 ha), which was necessary to find sufficient lizards to develop indices. However, like Degenhardt, Tollestrup only had a relative estimate of the number of lizards in an area with which to compare census counts. Thus, Tollestrup tested the efficacy of a technique that estimated relative abundance of lizards (10-day census) by comparing those index values to another measure of relative abundance. Her technique has been widely used and accepted by government agencies and environmental consultants in California without the requisite testing necessary to validate the technique.

The potential value of short-duration censuses for monitoring lizard populations at many sites and for relatively little cost motivated us to test this method using

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[2] Williams, D.F., D.J. Germano, and W. Tordoff, III. 1993. Population studies of endangered kangaroo rats and blunt-nosed leopard lizards in the Carrizo Plains Natural Area, California. California Department of Fish and Game, Nongame Bird and Mammal Section, Report 93-01, Sacramento, California, USA.

[3] Tollestrup, K. 1976. A standardized method of obtaining an index of densities of blunt-nosed leopard lizards, *Crotaphytus silus*. Unpublished report to U.S. Fish and Wildlife Service, Contract 14-16-0001-5793RF, Sacramento, California, USA.

complete demographic data that we gathered on blunt-nosed leopard lizards from 1989 to 1994 (Williams et al.[2] 1993, Germano et al. 1994, Germano and Williams, unpublished data). The purpose of our study is to compare abundance indices of blunt-nosed leopard lizards from 10-day counts (standard protocol for this species) to statistically rigorous abundance estimates from mark-resight of lizards during full-season studies. Because of different seasons of activity, we separately compared measures of abundance of the adult/yearling cohort and the hatchling cohort.

METHODS

We studied populations of blunt-nosed leopard lizards on the Elkhorn Plain, San Luis Obispo County, California (Germano et al. 1994). The Elkhorn Plain is an elevated bench (685–765 m) between the Temblor Range to the east and the lower elevation (about 600 m) Carrizo Plain to the west. The Elkhorn Plain represents foothill habitat of the blunt-nosed leopard lizard, although the greatest portion of its range historically was in the flat floor of the San Joaquin Valley (Montanucci 1965).

Two 8.1-ha plots were established during April 1988. One plot was located within the fenced Elkhorn Plain Ecological Reserve and the other was established approximately 2.4 km northwest of the reserve. Each plot was 277 x 293 m, with 16 census lines spaced 18.3 m apart following the standard design of plots recommended by Tollestrup[3] (1976). Both plots contained approximately the same amounts of shrubs, wash, roads, and soil types.

We conducted full-season censuses of blunt-nosed leopard lizards from 1990 to 1994. Each plot was walked in the morning or early afternoon during temperatures when lizards are most active (24–35°C, 1.27 cm above soil surface; 30–41°C, 1.27 cm below soil surface). We noosed unmarked lizards and noted capture location, determined sex, and applied a numeral to the dorsal surface with a felt-tipped pen. We took standard body measurements and weighed each lizard to the nearest gram. We injected a passive integrated transponder (PIT) subdermally to mark individuals permanently (Germano and Williams 1993). Besides capturing unmarked lizards, we kept counts of all blunt-nosed leopard lizards seen each day of censusing. In this way, we knew the total number of lizards we saw each day as well as the number of lizards we had marked.

Population size of blunt-nosed leopard lizards on each plot was determined using a Monte Carlo simulation method (Minta and Mangel 1989). This method estimates the total number of lizards on a plot by determining what percentage of the unmarked lizards that were recorded during the season actually were unmarked and not marked lizards that were misidentified (each day some lizards were too wary to catch or identify positively) by using capture frequencies of marked and unmarked lizards. The total abundance estimate for this method is the sum of the estimate of unmarked lizards from the simulation added to the total number of marked lizards. This method is especially useful for capture-resight data of the kind gathered during lizard censuses. We followed the recommendations of Minta and Mangle (1989) for determining upper and lower stopping limits for estimating abundance of unmarked lizards. The simulation was run 10,000 times.

We assumed that the population size estimated using full-season data accurately reflected true abundance. Adult abundance was estimated using full-season data each year (April through July). Hatchling abundance was estimated using full-season data collected from first emergence (July or August) to last activity for the year (October). Hatchling abundance was not estimated in 1990 because no reproduction occurred that year (Germano et al. 1994).

We regressed estimated abundance of blunt-nosed leopard lizards from full-season data on results of four types of 10-day censuses: 1) counts during 10 consecutive days of censusing, 2) counts during 10 random days of censusing, 3) number of lizards marked during 10 consecutive days of censusing, and 4) population size estimated from 10 consecutive days of censusing. Population size for 10 days of censusing was estimated in the same manner as full-season data, but only used the number of marked and unmarked lizards found on 10 consecutive d. For the adult/yearling cohort, counts were made during peak activity: late May–early June for consecutive counts and May–June for random counts. Hatchlings were counted during their period of peak activity: August for consecutive counts and late July–early September for random counts. A best subset regression (Analytical Software 1994) was used to determine the best combination of censuses for predicting mark-resight abundance estimates separately for the adult/yearling cohort and hatchling cohort.

RESULTS

Abundance of adult blunt-nosed leopard lizards was lowest in 1990 and steadily rose to a peak in 1993 on both plots (Table 1). Abundance of adults on both plots decreased in 1994. Abundance of hatchlings peaked in 1992 on one plot and in 1993 on the other plot; no lizards hatched in 1990 (Table 1). Blunt-nosed leopard lizards were consistently more abundant on the plot in the Elkhorn Plain Ecological Reserve than on the plot northwest of the reserve.

All four measures of population size for 10-day censuses were significantly (P <0.001) related to full-season abundance estimates for the adult/yearling cohort (Fig. 1) and hatchling cohort (Fig. 2). For both cohorts, the highest r^2 values were for random counts of lizards, although other measures of abundance were nearly as well related to full-season estimates (Table 2). When all four census types were combined, the linear combination was significant for the adult/yearling cohort ($F = 57.89$; df = 4, 9; $P = 0.0002$) and for the hatchling cohort ($F = 49.49$; df = 4, 7; $P = 0.0045$). Based on R^2 values, a two-factor model using consecutive counts and the number of marked lizards, and a three-factor model using consecutive counts, random counts, and the number of marked lizards were best for predicting full-season abundance of the adult/yearling cohort (Table 3). For the hatchling cohort, the highest R^2 value was for a three-factor model using consecutive counts, number of marked lizards, and 10-day population estimates. For both adult/yearlings and hatchlings, all multiple-factor models were only slightly better than random counts alone at predicting abundance (Table 3).

Table 1. Mark-resight estimates of abundance from full-season censuses (adult/yearlings, April–July; hatchlings, July–October) and abundances from four 10-day census methods for blunt-nosed leopard lizards in foothill habitats. Paired numbers are for plots 1 and 2. Plot 1 was in the Elkhorn Plain Ecological Reserve; Plot 2 was about 2.4 km northwest of the reserve.

Year	Full-season abundance estimate	Consecutive count	10-day census measure: Random count	10-day census measure: Number marked	Population estimate
			Adult/yearlings		
1990	14/11	16/13	25/10	6/6	20/19
1991	46/28	42/24	40/26	17/12	26/28
1992	67/45	61/15	64/32	28/10	36/11
1993	164/73	204/71	183/78	70/28	103/58
1994	110/42	159/121	152/86	53/36	74/38
			Hatchlings		
1990	0/0[a]				
1991	79/48	70/35	48/38	36/16	50/27
1992	219/107	256/58	255/106	147/43	197/93
1993	273/77	307/108	245/68	129/30	173/37
1994	74/23	56/31	70/24	23/14	33/19

[a] No blunt-nosed leopard lizards hatched in 1990 (Germano et al. 1994).

Table 2. Regression equations, r^2, F values, and probabilities for regressions of full-season mark-resight abundance estimates of blunt-nosed leopard lizards in foothill habitats on four other measures of abundance: counts during 10-day censuses (consecutive and random), number of lizards marked in 10 consecutive days of censusing, and population size estimates from 10-day censuses. For adults/yearlings, df = 1, 9; for hatchlings, df = 1, 7.

10-d census type	Equation	r^2	F	P
	Adults/Yearlings			
Consecutive count	Abundance = 14.6 + 1.28(Count)	0.80	32.2	0.0005
Random count	Abundance = 6.12 + 0.77(Count)	0.90	71.8	<0.0001
Marked	Abundance = 4.89 + 2.07(No. Marked)	0.89	63.8	<0.0001
Population estimate	Abundance = -3.31 + 1.53(Pop. Est.)	0.89	62.3	<0.0001
	Hatchlings			
Consecutive count	Abundance = 20.9 + 0.80(Count)	0.94	94.7	0.0001
Random count	Abundance = 13.7 + 0.93(Count)	0.95	114.4	<0.0001
Marked	Abundance = 25.4 + 1.59(No. Marked)	0.92	66.8	0.0002
Population estimate	Abundance = 18.7 + 1.19(Pop. Est.)	0.89	60.3	0.0002

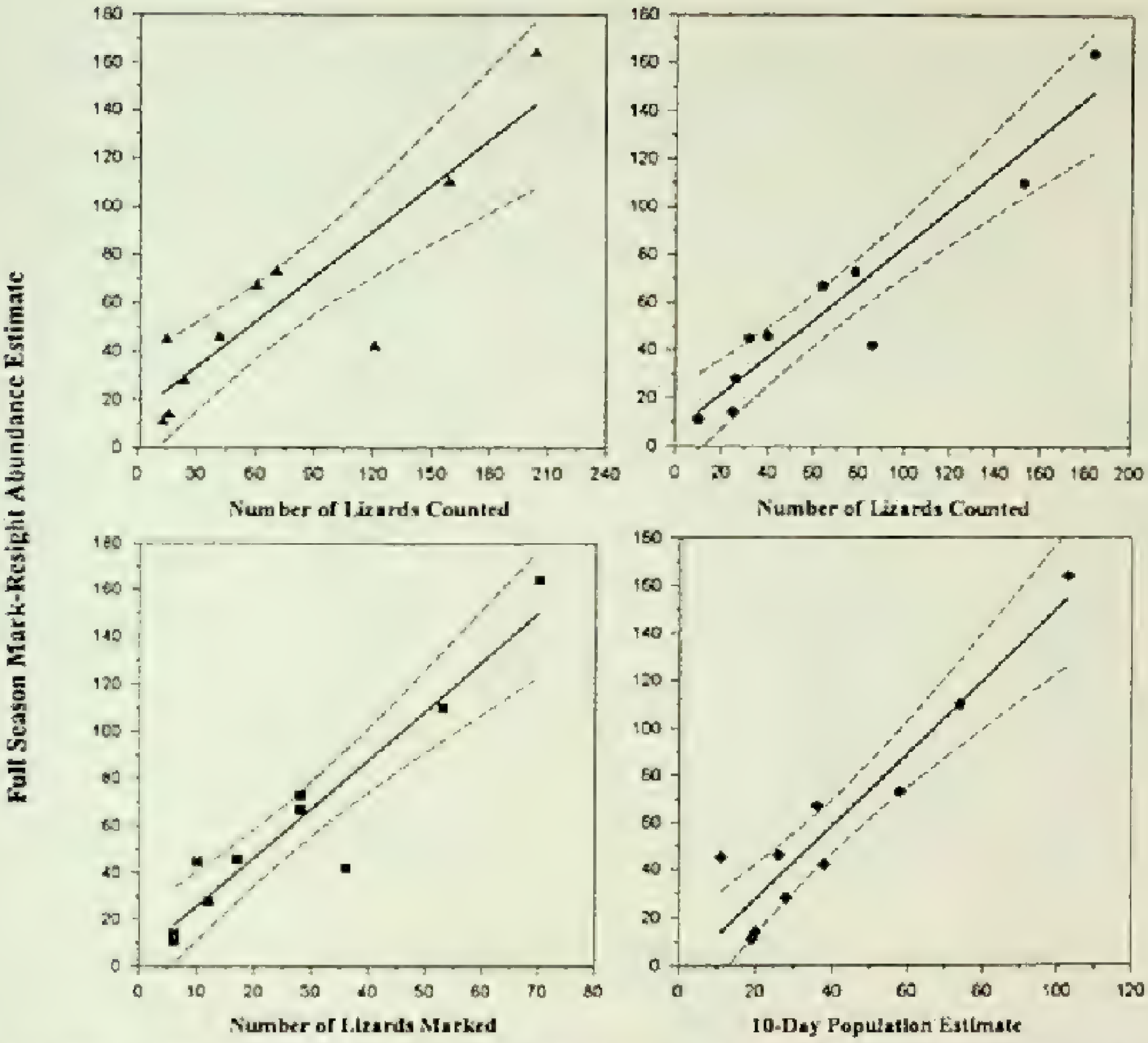

Figure 1. Relationships between full-season abundance estimates of adult/yearling blunt-nosed leopard lizards and abundance measures from 10-day censuses. Relationships are for counts during 10 consecutive days of censusing (triangles), counts during 10 random days of censusing (dots), number of lizards marked in 10 consecutive days of censusing (squares), and abundance estimated using mark-resight data gathered during 10 consecutive days of censusing (diamonds). Dashed lines are 95% confidence intervals.

DISCUSSION

We found that the 10-day census of blunt-nosed leopard lizards was a satisfactory technique to index abundance. We found significant relationships between four measures of abundance from 10-day censuses of both the adult/yearling cohort and the hatchling cohort to independent estimates of abundance using mark-resight data collected during full-season censuses. Although all four measures of 10-day censuses were strongly related to the mark-resight estimates, highest values were those that used counts of lizards made during 10 random days of censusing during peak activity of blunt-nosed leopard lizards. Linear combinations of the four measures gave a slightly better fit to mark-resight abundance estimates than random censuses, but, given the amount of work involved, we do not believe it is worthwhile to capture and mark lizards during 10-day censuses.

Ten-day censuses also may be useful for monitoring abundance of other lizard species in California, especially conspicuous species such as the long-nosed leopard lizard, *Gambelia wislizenii*; the side-blotched lizard, *Uta stansburiana*; various species

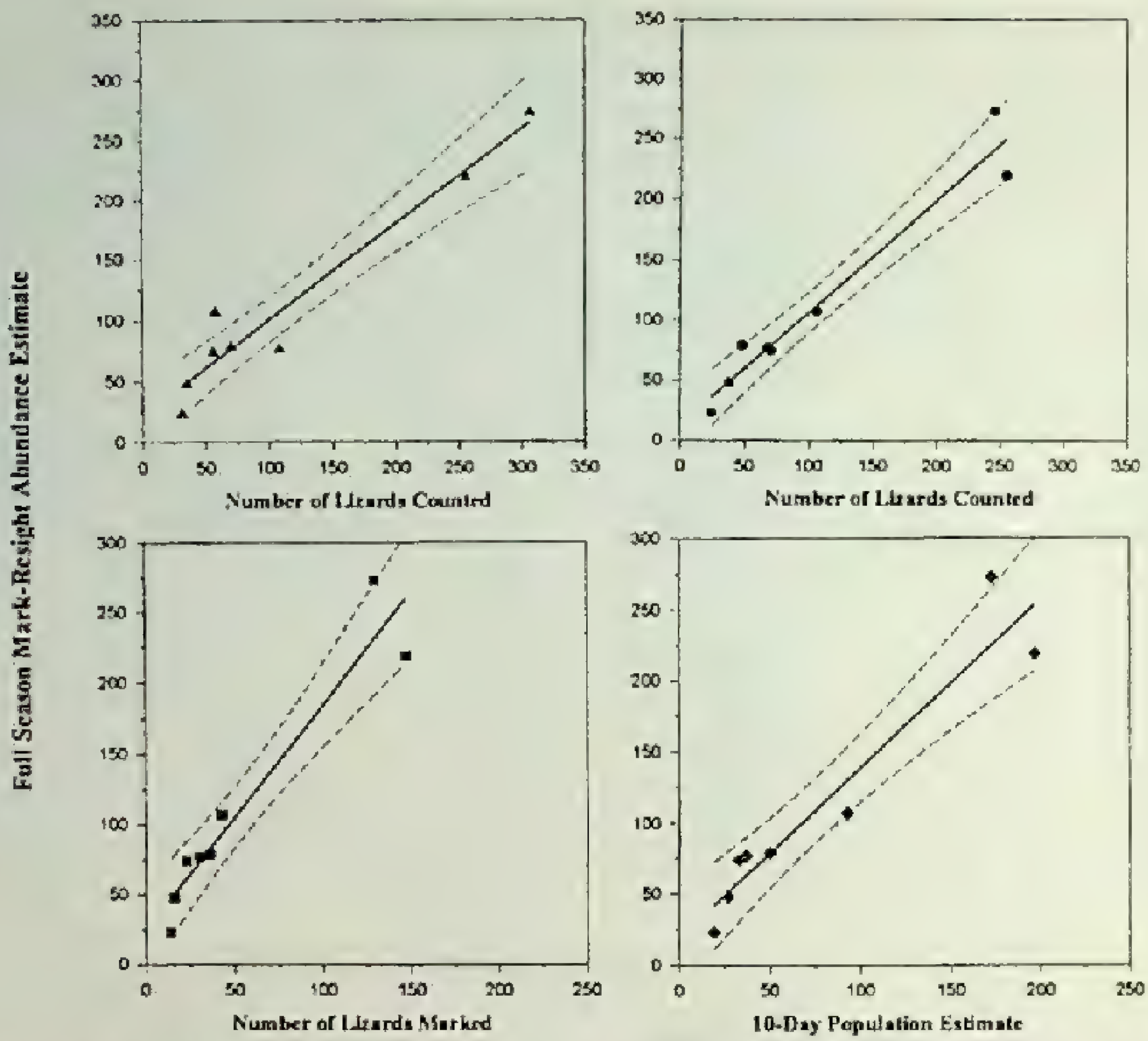

Figure 2. Relationships between full-season abundance estimates of hatchling blunt-nosed leopard lizards and abundance measures from 10-day censuses. Relationships are for counts during 10 consecutive days of censusing (triangles), counts during 10 random days of censusing (dots), number of lizards marked in 10 consecutive days of censusing (squares), and abundance estimated using mark-resight data gathered during 10 consecutive days of censusing (diamonds). Dashed lines are 95% confidence intervals.

of fence lizards, *Sceloporus* spp., and whiptail lizards, *Cnemidophorus* spp.; the zebra-tailed lizard, *Callisaurus draconoides*; and the desert iguana, *Dipsosaurus dorsalis*. However, the relationship between counts and true abundance should be estimated for each species. This requires that abundance be determined for either many plots or for several years, or both, and of necessity means marking a large portion of the population. Once the relationship is established between counts and abundance, 10-day censuses can be used at other sites as long as the vegetation structure is similar. In this light, we caution against using our predictive equations of counts and abundance beyond the foothill habitat of blunt-nosed leopard lizards.

In many research studies, it is both desirable and necessary to capture individual lizards. In these cases, 10-day censuses are not useful. However, other research studies and some environmental work only require a good index of abundance. When only population indices are needed for blunt-nosed leopard lizards, we suggest that 10-day censuses be used.

Table 3. Best subset regression models for predicting mark-resight abundance estimates of blunt-nosed leopard lizards in foothill habitats from unforced independent variables A (counts during 10-day consecutive censuses), B (counts during 10-day random censuses), C (number of lizards marked in 10 consecutive days of censusing), and D (population estimates made using data from 10-day censuses).

Adults/Yearlings		Hatchlings	
Variables in Model	R^2	Variables in Model	R^2
B	0.90	B	0.95
C	0.89	A	0.94
D	0.89	C	0.92
A	0.80	D	0.89
A, C	0.96	A, B	0.97
A, B	0.94	A, D	0.97
B, D	0.91	A, C	0.95
A, B, C	0.98	A, C, D	0.98
A, C, D	0.97	A, B, C	0.97
A, B, D	0.95	A, B, D	0.97
A, B, C, D	0.98	A, B, C, D	0.99

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SURVEY OF SMALL FISHES AND ENVIRONMENTAL CONDITIONS IN MUGU LAGOON, CALIFORNIA, AND TIDALLY INFLUENCED REACHES OF ITS TRIBUTARIES

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From September to November 1993, 7,641 fishes representing 16 species were captured at 14 sampling sites in or immediately upstream from Mugu Lagoon, a 1,073-ha estuarine lagoon in Ventura County, California. Topsmelt, *Atherinops affinis*; California killifish, *Fundulus parvipinnis*; arrow goby, *Clevelandia ios*; and western mosquitofish, *Gambusia affinis*, were the most abundant species. Cluster analysis and association analysis using fish presence-absence data identified two major fish species assemblages, one typified by marine species and the other by freshwater species. Simple discriminant analysis of selected environmental variables indicated that salinity contributed the most towards separating the two assemblages, followed by temperature and dissolved oxygen. These findings support the generally accepted view that estuarine fish species assemblages are structured primarily by the tolerance of individual species to environmental variables such as salinity and temperature.

INTRODUCTION

Although nearly all lagoons and estuaries along the southern California coast have been extensively modified by human activities, Mugu Lagoon is unique in that public access has been restricted since 1946 when the Point Mugu Naval Air Station (NAS) was established (Onuf[1] 1987). The relative isolation of this lagoon from recreational pressures has created a semi-protected environment for fish and wildlife. However, much of the shoreline of Mugu Lagoon is bordered by naval facilities (e.g., runways, hangers, workshops, storage bunkers, fuel depots, and military housing) or roads and its watershed has been extensively modified by agriculture and urbanization.

The environmentally degraded condition of Mugu Lagoon and its tributaries was noted in the 1994 California Water Quality Assessment, which described Calleguas Creek, Revolon Slough, and the lagoon as having impaired water quality

[1]Onuf, C.P. 1987. The ecology of Mugu Lagoon, California: An estuarine profile. Biological Report 85(7.15), U.S. Fish and Wildlife Service, Washington, D.C., USA.

(SWRCB[2] 1994). Results from the California Toxic Substances Monitoring Program also indicated that fishes in the lagoon and its tributaries contain among the highest concentrations of arsenic, silver, DDT, and methoxychlor measured in California (Rasmussen[3] 1995 and earlier reports). In addition, dacthal, toxaphene, chlordane, and endosulfan have been measured in fishes from the lagoon and its tributaries at inordinately high concentrations (Rasmussen[3] 1995 and earlier reports).

Amid growing concern over the possible effects that degraded environmental conditions might be having on fishes in Mugu Lagoon, the U.S. Department of the Navy contracted with the Research Division of the U.S. Fish and Wildlife Service (now the Biological Resources Division of the U.S. Geological Survey) to inventory the fishes, with special emphasis on small species inhabiting shallow (<1-m-deep) water. The emphasis on small species in shallow water resulted from a desire to maximize the likelihood of capturing tidewater goby, *Eucyclogobius newberryi*, an endangered fish that historically occurred in the lagoon (Swift et al. 1989). Specific objectives of the study were to (i) inventory fish species that inhabit Mugu Lagoon and tidally influenced reaches of its tributaries, (ii) determine if fishes were segregated into distinctive species assemblages, and (iii) determine if species assemblages were delimited by water quality or other physicochemical variables.

STUDY AREA

Fourteen sampling sites were established in Mugu Lagoon or in tidally influenced reaches of its tributaries (Fig. 1). Mugu Lagoon is the largest estuarine lagoon in southern California (terminology after Davies 1980), covering 1,073 ha of open water and tidal or intertidal wetlands (T. Keeney, Point Mugu NAS, California, pers. comm.). The lagoon is contained entirely within the NAS at Point Mugu in Ventura County, about 16 km southeast of the City of Oxnard. Although environmental conditions are generally marine dominated, the lagoon receives freshwater inflows from Calleguas Creek, Revolon Slough, Oxnard drains #2 and #3, and other sources (including runoff from the Santa Monica Mountains) in its 1,036-km^2 watershed. Calleguas Creek and Revolon Slough also provide a continuous flow of agricultural wastewater and tertiary-treated sewage plant return water to the lagoon.

Sampling for this study occurred on 13–16 September, 19–21 October, and 15–18 November 1993. In addition, a preliminary trip was made on 2–3 August 1993 to refine protocols for fish collections and to resolve other logistical problems.

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[2] SWRCB (State Water Resources Control Board). 1994. Draft water quality assessment (WQA). State Water Resources Control Board, California Environmental Protection Agency, Sacramento, California, USA.

[3] Rasmussen, D. 1995. Toxic substances monitoring program: 1992-93 data report. Publication No. 95-1WQ, State Water Resources Control Board, California Environmental Protection Agency, Sacramento, California, USA.

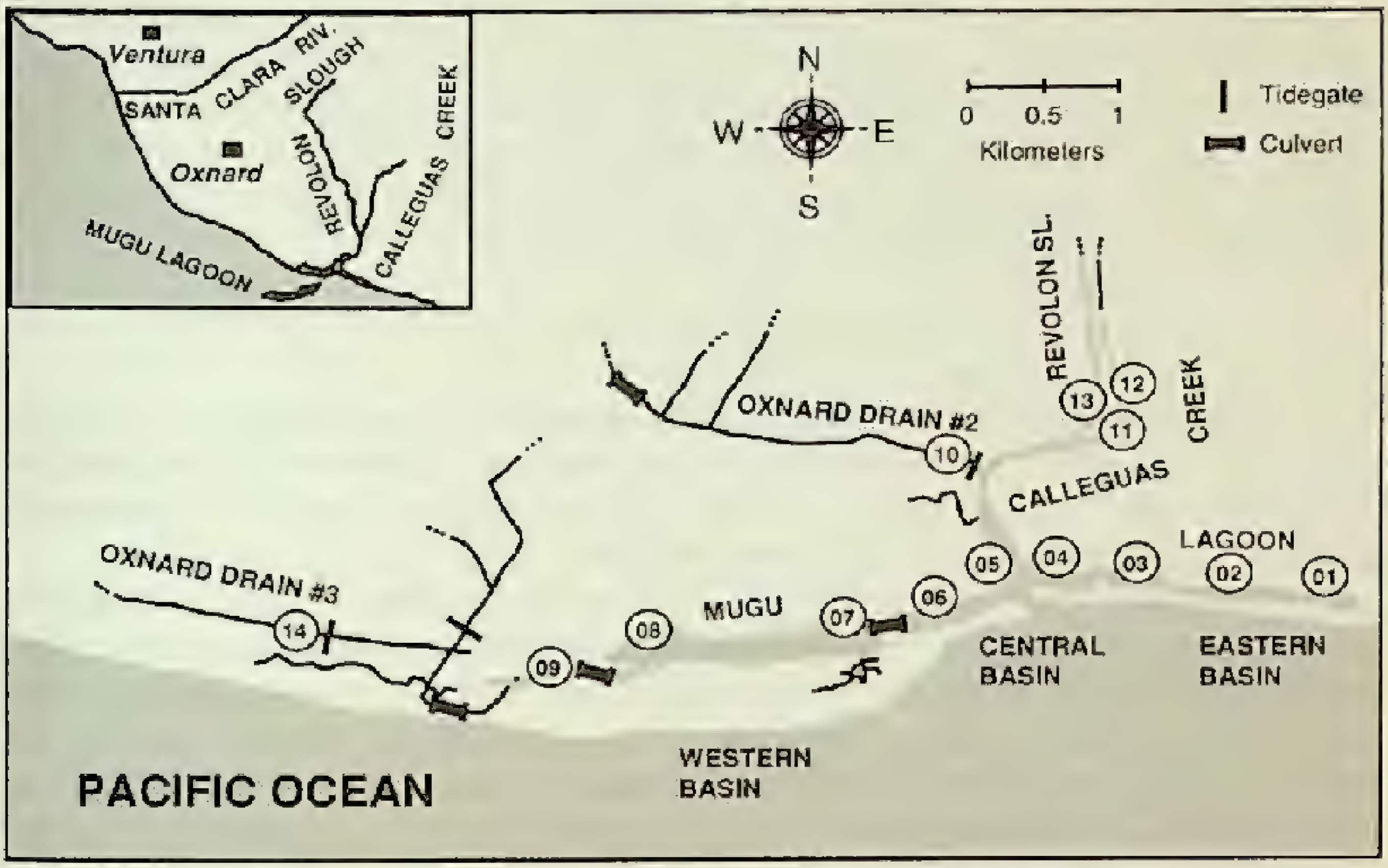

Figure 1. Map of the study area showing the locations of 14 sampling sites.

METHODS

Fishes

On each of the three sampling trips, fishes were collected during daylight at individual sites by making five "blind" throws with a 1.8-m radius throw net. In addition, a 15-m-long haul was made either perpendicular or parallel to shore with a 5.5-m-long (1.8-m-deep) bag seine. All nets were constructed with 3.2-mm-square mesh.

Immediately after collection, fishes were identified to species, counted, and most were then released alive at the capture site. Voucher specimens and dead or injured fishes were preserved in 10% formalin for verification of field identifications with taxonomic keys by Eddy (1969), Miller and Lea (1972), Moyle (1976), and Eschmeyer and Herald (1983). In addition, identifications of several species were provided by W.N. Eschmeyer, Department of Ichthyology, California Academy of Sciences, San Francisco, California (CAS Accession No. 1994-III:3; specklefin midshipman, *Porichthys myriaster*; striped mullet, *Mugil cephalus*; barred sand bass, *Paralabrax nebulifer*; and deepbody anchovy, *Anchoa compressa*).

Relative abundance of fish species captured by throw net and bag seine was defined as follows: very abundant (V), the number of individuals captured by a given gear type at a particular site exceeded 10% of the total catch (all species combined) by that gear type from all sites combined; abundant (A), the number of captured individuals was >1% but ≤10% of the total catch; common (C), the number of captured individuals was >0.1% but ≤1% of the total catch; rare (R), the number of captured

individuals was >0% but ≤0.1% of the total catch; and absent or not captured (0), no individuals were captured. Certain fishes (e.g., flatfish and some gobies) may have been inadequately sampled by the two gear types because these taxa can burrow into the substrate. Also, large and fast swimmers (e.g., sharks, rays, and guitarfish) that inhabited open-water areas or deep, swift-flowing channels were probably under-represented in the catches.

Physicochemical Variables

Physicochemical variables were measured at each sampling site near where fishes were collected. Water quality measurements were taken immediately prior to fish collections. Sediment samples were collected only during November, after water quality measurements, but prior to fish collection.

Water temperature, pH, dissolved oxygen concentration, salinity, and specific conductance were routinely measured 15–30 cm below the surface with a Hydrolab Reporter Multiprobe attached to a Hydrolab Surveyor 2 Display Logger (Hydrolab Corporation, Austin, Texas[4]). In addition, water samples were collected in 1-liter brown plastic bottles, then cooled in an ice chest or refrigerator until turbidity was measured (usually within 4–6 days) with a portable LaMotte Model 2008 turbidimeter (LaMotte Company, Chestertown, Maryland[4]).

Sediment samples were collected in clean 1-liter round plastic containers with leak-proof "snap-on" lids. The sediment samples were obtained as follows: 1) the surficial layer of sediment (to a soil depth of 2–3 cm) was gently scooped into each plastic container; 2) the filled container was slowly raised out of the water and most of the overlying water was poured off (being careful not to agitate the sample); and 3) a lid was placed on the container and the container was cooled on ice. After transport to the laboratory, samples were frozen at −10°C until analyzed for sediment particle size distribution (within 3 months).

Sediment particle size distribution was determined by sifting the thawed samples through a series of standard sieves (mesh diameters of 31.5, 9.5, 2.0, 1.4, and 0.053 mm), then measuring the dry weights of each sediment fraction and the filtrate passing through the 0.053-mm sieve. For this study, sediment particles ≤0.053 mm are referred to as silt; >0.053–1.4 mm, as sand; >1.4–2.0 mm, as coarse sand; >2.0–9.5 mm, as gravel; and >9.5–31.5 mm, as pebble. Schoklitsch's sediment factor (s) was calculated with a graphical procedure described by Bogardi (1974). By definition, $s = a/b$, where a is the area above the granulometric curve and b is the area below the granulometric curve. The granulometric curve for each sediment sample was drawn by plotting the diameter of the sediment fractions (i.e., 0.053, 1.4, 2.0, 9.5, and 31.5 mm) on the x-axis and the cumulative percentages of their dry weights on the y-axis. According to Bogardi (1974), the value for s (which can assume any arbitrary positive value) increases as sediment material becomes coarser.

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[4] Reference to trade names does not imply an endorsement by the U.S. Geological Survey or the California Department of Fish and Game.

Data Analysis

Except as noted below, nonparametric procedures (e.g., chi-square test for homogeneity, Kruskal-Wallis test, Spearman rank correlation) were used for statistical comparisons. A multiple-means comparison test described by Conover (1980), where pairwise t-tests (equivalent to Fisher's least-significant difference test in the case of equal cell sizes) were performed with ranked values, was used only if significant ($P < 0.05$) chi-square values were computed by the Kruskal-Wallis test.

Green and Vascotto (1978) recommended using cluster analysis to identify species-assemblage groups, followed by discriminant analysis to determine if the composition of species-assemblage groups is controlled by environmental factors. However, there are no satisfactory methods for determining the number of clusters (groups) (Hawkins et al. 1982). According to Ludwig and Reynolds (1988), the identification of specific groups or communities by cluster analysis is often done subjectively. As a general guideline, Ludwig and Reynolds (1988) recommended that the clusters not be divided so finely that the analysis results in a large number of fragmented and uninterpretable groups. Other classification techniques such as association analysis can be used to gain concordance with the findings from cluster analysis. During this study, several methods of cluster analysis (e.g., Ward's minimum variance, single linkage, average linkage) and association analysis were performed with presence-absence data for fishes to group the sampling sites, whereas simple discriminant analysis was performed with ranked values of environmental variables to test for statistical differences among the grouped sites.

With two exceptions, all statistical computations were conducted with SAS software under Windows, version 6.10 (SAS Institute Inc., Cary, North Carolina). Computations for association analysis and simple discriminant analysis were conducted with BASIC programs written by Ludwig and Reynolds (1988).

RESULTS

A total of 7,641 fish representing 16 species from 14 families was captured during this study (Table 1). Six additional species were either captured only during the preliminary trip in August (yellowfin goby, *Acanthogobius flavimanus*), captured at other localities in the study area (goldfish, *Carassius auratus*, and striped mullet), or were observed but not captured (leopard shark, *Triakis semifasciata*; bat ray, *Myliobatis californica*; and shovelnose guitarfish, *Rhinobatos productus*). Despite an intensive effort to collect tidewater goby, this species was not captured.

Ten fish species were captured by throw net and 16 species were captured by bag seine (Table 1). Proportions (percentages) of captured fish species varied significantly between gear types ($\chi^2 = 178.2$, df = 15, $P = 0.001$). When catches in the two gears were compared, topsmelt, *Atherinops affinis*; Pacific staghorn sculpin, *Leptocottus armatus*; and western mosquitofish, *Gambusia affinis*, accounted for higher proportions of the throw net catch, whereas California halibut, *Paralichthys californicus*; arroyo chub, *Gila orcutti*; California killifish, *Fundulus parvipinnis*; shiner perch, *Cymatogaster aggregata*; and arrow goby, *Clevelandia ios*, accounted for higher proportions of the bag seine catch (Table 1).

Table 1. List of fish species and numbers of individuals captured by throw net and bag seine in Mugu Lagoon and its tributaries, September-November 1993.

Taxon	Number of fish (% of catch) Throw net	Bag Seine
Atherinidae		
Topsmelt, *Atherinops affinis*	1269 (84)	4406 (72)
Batrachoididae		
Specklefin midshipman, *Porichthys myriaster*	0 (0)	2 (<1)
Bothidae		
California halibut, *Paralichthys californicus*	0 (0)	9 (<1)
Carcharhinidae		
Gray smoothhound, *Mustelus californicus*	2 (<1)	3 (<1)
Clupeidae		
Threadfin shad, *Dorosoma petenense*	0 (0)	1 (<1)
Cottidae		
Pacific staghorn sculpin, *Leptocottus armatus*	15 (1)	21 (<1)
Cyprinidae		
Common carp, *Cyprinus carpio*	0 (0)	1 (<1)
Arroyo chub, *Gila orcutti*	1 (<1)	64 (1)
Cyprinodontidae		
California killifish, *Fundulus parvipinnis*	137 (9)	963 (16)
Embiotocidae		
Shiner perch, *Cymatogaster aggregata*	0 (0)	70 (1)
Engraulidae		
Deepbody anchovy, *Anchoa compressa*	1 (<1)	1 (<1)
Gobiidae		
Arrow goby, *Clevelandia ios*	37 (2)	444 (7)
Longjaw mudsucker, *Gillichthys mirabilis*	0 (0)	2 (<1)
Pleuronectidae		
Diamond turbot, *Hypsopsetta guttulata*	2 (<1)	14 (<1)
Poeciliidae		
Western mosquitofish, *Gambusia affinis*	45 (3)	129 (2)
Serranidae		
Barred sand bass, *Paralabrax nebulifer*	1 (<1)	1 (<1)

Occurrence, Relative Abundance, and Fish Species Assemblages

Topsmelt was the most ubiquitous species (found at all 14 sites), followed by California killifish (13 sites), arrow goby (nine sites) and Pacific staghorn sculpin (nine sites) (Table 2). The remaining species were collected at five or fewer sites.

Topsmelt, California killifish, shiner perch, arrow goby, and western mosquitofish were "very abundant" or "abundant" at one or more sites (Table 2). Topsmelt dominated the catch at most sites. By comparison, California killifish were abundant at one site each in the eastern, central, and western basins of Mugu Lagoon and in Calleguas Creek. Shiner perch were abundant only in Oxnard Drain #2. Arrow goby were abundant at two sites in the eastern basin. Western mosquitofish were abundant in the uppermost site on Calleguas Creek (Site 12) and in Oxnard Drain #3.

Table 2. Relative abundance of fish captured in Mugu Lagoon and its tributaries, September–November 1993. Data for species are listed as follows: throw net, bag seine. Codes: V, very abundant; A, abundant; C, common; R, rare; 0, absent or not captured; –, no data. See text for details.

Site[a]	Topsmelt	Specklefin midshipman	California halibut	Gray smoothhound	Threadfin shad	Pacific staghorn sculpin	Common carp	Arroyo chub
01	A,A	0,0	0,0	0,0	0,0	0,0	0,0	0,0
02	C,A	0,0	0,0	0,0	0,0	0,0	0,0	0,0
03	A,A	0,0	0,0	0,0	0,0	R,0	0,0	0,0
04	A,A	0,0	0,0	0,0	0,0	R,R	0,0	0,0
05	A,A	0,0	0,R	0,R	0,0	0,0	0,0	0,0
06	C,C	0,0	0,R	0,0	0,0	R,0	0,0	0,0
07	C,R	0,0	0,R	0,0	0,0	R,0	0,0	0,0
08	A,A	0,0	0,0	R,R	0,0	C,C	0,0	0,0
09	V,V	0,R	0,0	0,0	0,0	0,0	0,0	0,0
10	C,A	0,0	0,R	0,0	0,0	R,R	0,0	0,0
11	A,A	0,0	0,0	0,0	0,0	0,0	0,0	R,C
12	C,A	0,0	0,0	0,0	0,0	0,R	0,R	0,C
13	C,A	0,0	0,0	0,0	0,R	R,R	0,0	0,R
14	C,–	0,–	0,–	0,–	0,–	C,–	0,–	0,–

Site[a]	California killifish	Shiner perch	Deepbody anchovy	Arrow goby	Longjaw mudsucker	Diamond turbot	Western mosquitofish	Barred sand bass
01	R,C	0,0	R,0	C,R	0,0	0,R	0,0	0,0
02	C,A	0,0	0,0	C,A	0,0	R,0	0,0	0,R
03	0,C	0,0	0,0	C,A	0,R	0,0	0,0	0,0
04	A,A	0,R	0,0	0,C	0,0	0,R	0,0	0,0
05	0,R	0,0	0,0	0,C	0,0	0,R	0,0	0,0
06	0,R	0,0	0,0	0,C	0,0	R,C	0,0	0,0
07	R,R	0,0	0,0	R,C	0,0	0,0	0,0	0,0
08	R,C	0,0	0,0	0,C	0,0	0,0	0,0	0,0
09	A,A	0,0	0,0	0,R	0,0	0,0	0,0	R,0
10	0,R	0,A	0,0	0,0	0,R	0,0	0,0	0,0
11	C,A	0,R	0,0	0,0	0,0	0,0	0,R	0,0
12	C,R	0,0	0,0	0,0	0,0	0,0	0,A	0,0
13	0,C	0,R	0,R	0,0	0,0	0,0	0,R	0,0
14	0,–	0,–	0,–	0,–	0,–	0,–	A,–	0,–

[a] See Fig. 1 for locations of sampling sites.

According to Ward's minimum-variance method of cluster analysis with fish presence-absence data, the 14 sampling sites can be divided into two major groups: one composed of sites 01–10 and the other composed of sites 11–14 (Fig. 2). These two groups are also evident in single linkage, average linkage, and other agglomerative hierarchical clustering procedures. However, further subdivision of the two groups yielded inconsistent patterns among clustering procedures, indicating that the interpretive value was not improved by examining results from additional clustering cycles.

Association analysis performed with fish presence-absence data supported the finding from cluster analysis that the 14 sampling sites can be divided into two groups: sites 01–10 and sites 11–14 (Fig. 3). Western mosquitofish was the "divisor" species; i.e., it accounted for the largest number of significant chi-square values. In addition, western mosquitofish was negatively associated with arrow goby ($\chi^2 = 6.54$, df = 1, $P < 0.025$) and positively associated with arroyo chub ($\chi^2 = 5.61$, df = 1, $P < 0.025$).

The two groups of sites were populated by distinctly different assemblages of fishes. Sites 01–10 were characterized by marine fishes such as specklefin midshipman; California halibut; gray smoothhound, *Mustelus californicus*; arrow goby; longjaw mudsucker; diamond turbot; and barred sand bass (Table 2). By comparison, sites 11–14 were characterized by freshwater fishes such as threadfin shad, common carp, arroyo chub, and western mosquitofish. Ubiquitous species occurring in both groups of sites included topsmelt, Pacific staghorn sculpin, California killifish, and shiner perch. Although deepbody anchovy was present in both groups of sites, its occurrence at Revolon Slough (a freshwater site) coincided with temporary intrusion of saltwater during high tide.

Environmental Variables

Water quality variables varied as follows: temperature, 9.8–24.9°C; pH, 7.4–8.5; dissolved oxygen concentration, 1.9–17.8 mg/liter; salinity, 0.5–34.3‰; and turbidity, 0.2–29.4 nephelometric units (NTUs) (Table 3). However, only salinity differed significantly among sites. In general, sites located within the eastern, central, and

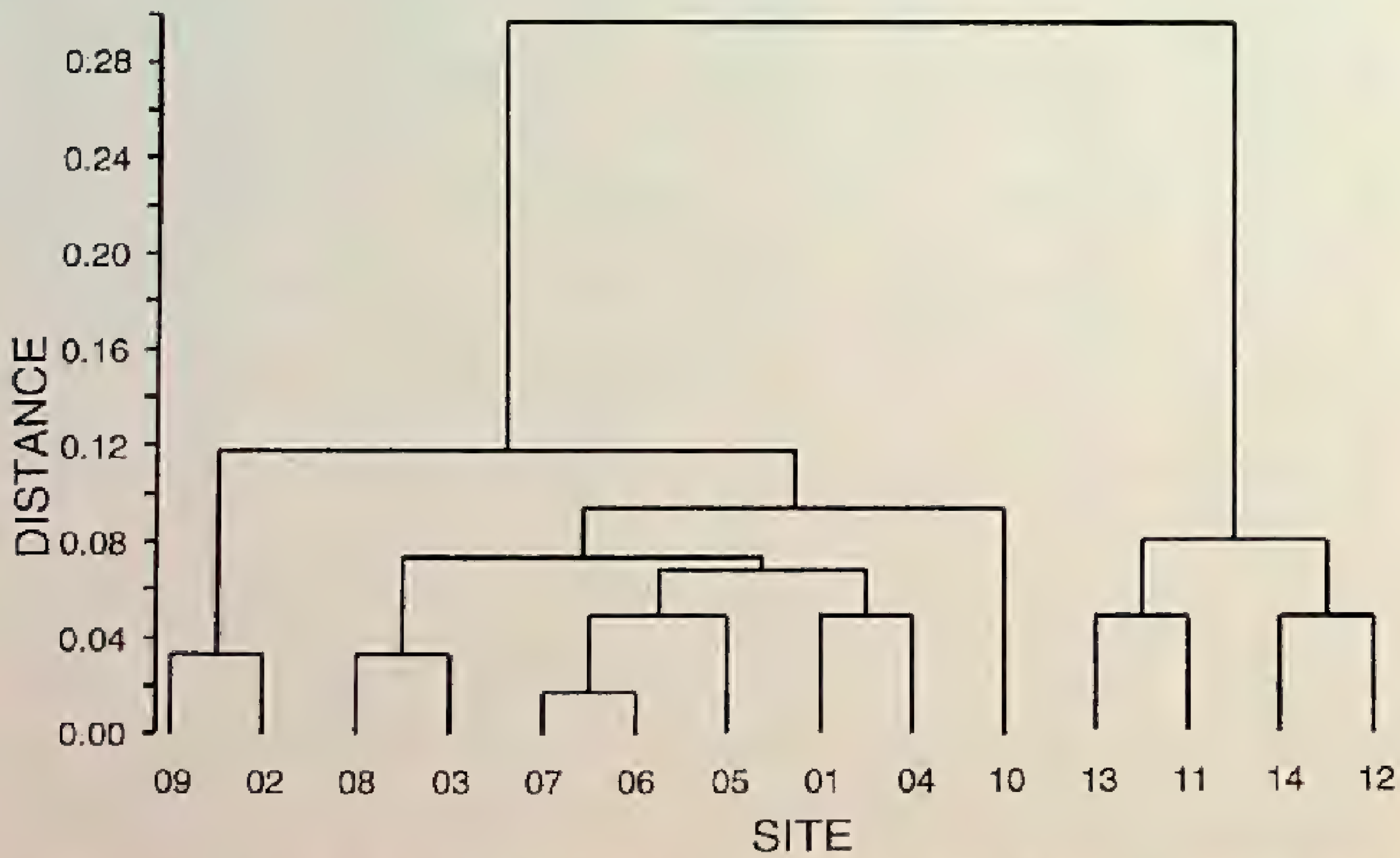

Figure 2. Dendrogram for cluster analysis (Ward's minimum-variance method) of 14 sampling sites in Mugu Lagoon and its tributaries. Cluster analysis was performed with presence-absence data for 16 fish species.

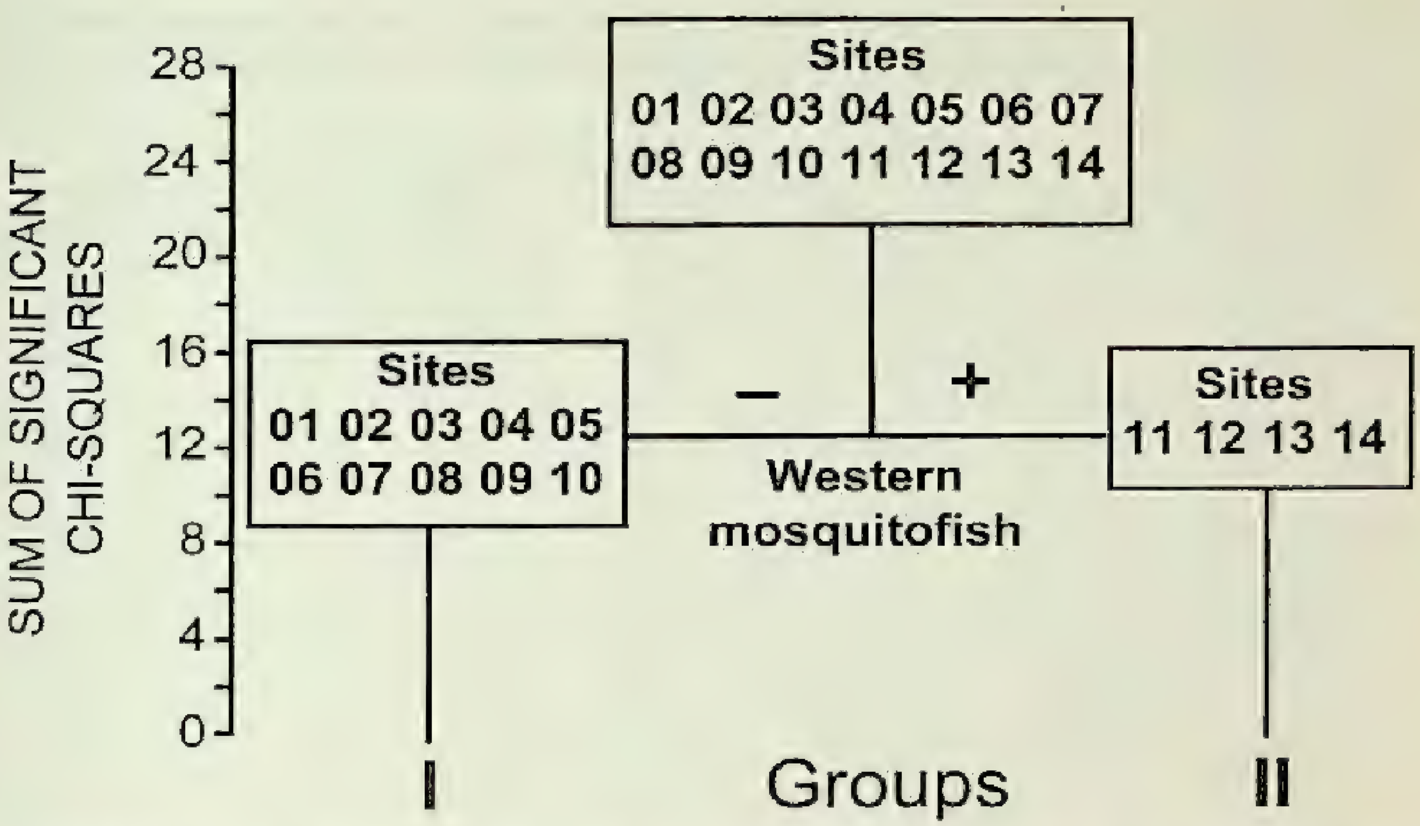

Figure 3. Division of 14 sampling sites in Mugu Lagoon and its tributaries using an association analysis based on the presence-absence of 16 fish species.

western basins of Mugu Lagoon were more saline than sites located in Calleguas Creek, Revolon Slough, and the Oxnard drains.

The particle sizes of bottom sediments from most sites were dominated by sand and silt (Table 4). Schoklitsch's sediment factor varied over a narrow range from 0.019 (the fineness sediments) at Site 10 in Oxnard Drain #2 to 0.070 (the coarsest sediments) at Site 03 in the eastern basin.

The only significant correlations among environmental variables were between dissolved oxygen and pH ($r_s = 0.777$, df = 40, $P = 0.0001$) and between salinity and turbidity ($r_s = -0.589$, df = 40, $P = 0.0001$). Dissolved oxygen concentrations and pH were high in September and October, and low in November. The inverse relationship between salinity and turbidity resulted from high salinities and low turbidities in the eastern, central, and western basins and lower salinities and higher turbidities in Calleguas Creek, Revolon Slough, and Oxnard drains #2 and #3.

Relation Between Species Assemblages and Environmental Variables

According to simple discriminant analysis, significant differences existed among the abiotic characteristics of the two groups of sites (01–10 and 11–14) classified by cluster analysis and association analysis (Fig. 4). The multivariate distance (Mahalanobis D^2 statistic, 17.738) separating the two groups was significant ($F = 4.93$; df = 6, 7; $P = 0.0276$). Moreover, all 14 sites were correctly grouped according to whether they contained predominantly marine species or freshwater species. Salinity contributed the most towards the total multivariate distance (103.4%), followed by temperature (4.5%), and dissolved oxygen (2.2%). Schoklitsch's sediment factor

Table 3. Mean (range) water temperature(°C), pH, dissolved oxygen (DO, mg/l), salinity (‰), and turbidity (NTUs) in Mugu Lagoon and its tributaries, September–November 1993. N = 3 for all variables.

Site[a]	Temperature	pH	DO	Salinity[b, c]	Turbidity
01	17.5 (15.7–18.7)	7.9 (7.5–8.1)	7.8 (1.9–11.8)	32.5 (31.3–33.2)AB	5.6 (2.0–10.5)
02	16.9 (15.6–18.1)	7.8 (7.4–8.1)	5.9 (2.5–8.0)	32.9 (31.9–33.7)AB	2.5 (0.8–4.1)
03	18.0 (15.4–22.3)	8.0 (7.9–8.2)	9.4 (6.6–13.7)	32.1 (30.0–33.8)AB	3.2 (0.2–5.0)
04	17.0 (15.2–20.4)	8.0 (7.9–8.0)	8.7 (7.7–10.4)	32.1 (29.4–33.9)AB	3.6 (1.3–5.4)
05	15.4 (12.8–18.7)	7.8 (7.8–7.9)	8.2 (8.0–8.5)	26.1 (25.2–26.9)CD	9.2 (3.7–14.7)
06	16.0 (14.7–18.1)	7.9 (7.8–7.9)	7.7 (6.4–8.9)	33.7 (33.2–34.3)A	4.2 (2.8–6.2)
07	17.2 (15.7–18.9)	7.9 (7.8–8.1)	8.1 (5.8–10.6)	33.1 (32.6–33.5)AB	4.5 (3.8–5.0)
08	16.5 (11.1–22.3)	8.0 (7.9–8.1)	8.6 (6.7–11.7)	31.8 (30.3–33.1)BC	7.5 (3.8–12.7)
09	18.9 (15.7–24.9)	8.2 (7.7–8.5)	11.6 (3.7–16.9)	30.6 (28.1–33.3)BC	6.7 (4.6–8.2)
10	18.7 (14.9–21.0)	7.8 (7.5–8.0)	11.2 (7.4–17.8)	15.9 (13.8–19.6)DE	18.2 (9.4–29.4)
11	15.9 (14.1–19.2)	7.8 (7.6–7.9)	8.0 (6.8–8.9)	13.0 (0.6–22.6)DE	11.2 (3.1–22.6)
12	15.1 (12.6–18.6)	7.9 (7.9–7.9)	8.8 (7.6–9.8)	0.9 (0.5–1.7)E	12.7 (7.3–18.2)
13	16.1 (13.8–18.0)	7.9 (7.8–8.2)	8.6 (7.4–9.3)	17.6 (1.9–30.7)D	11.7 (4.6–19.2)
14	15.7 (9.8–19.2)	7.8 (7.5–8.0)	7.4 (6.3–9.4)	15.0 (14.2–15.6)DE	5.5 (2.0–11.2)
Kruskal -Wallis χ^2	5.970	5.922	5.868	33.252[d]	19.593

[a] See Fig. 1 for locations of sampling sites.

[b] Means followed by the same letter are not significantly different (P>0.05) according to Fisher's least significant difference test with ranked values.

[c] Specific conductance (mmhos/cm @ 25°C) can be estimated from salinity (‰) by the following equation: SC = 2.030963 + 1.478596(Salinity). N = 57, r^2 = 0.99.

[d] P <0.01.

did not contribute anything (0.0%), whereas pH (–2.6%) and turbidity (–7.6%) weakened the analysis.

DISCUSSION

Although water quality in Mugu Lagoon and its tributaries is classified as "impaired" (SWRCB[2] 1994) and aquatic biota from these waters are contaminated with among the highest concentrations of heavy metals and pesticides measured statewide (Rasmussen[3] 1995 and earlier reports), the fishes are similar in composition and relative abundance to those inhabiting lagoons and estuaries elsewhere along the southern California coast (Zedler[5] 1982, Horn and Allen 1985, Zedler and Nordby[6] 1986). During my study, the most abundant and widely distributed species

[5] Zedler, J.B. 1982. The ecology of southern California coastal salt marshes: A community profile. FWS/OBS-81/54, U.S. Fish and Wildlife Service, Biological Services Program, Washington, D.C., USA.

[6] Zedler, J.B. and C.S. Nordby. 1986. The ecology of Tijuana Estuary, California: An estuarine profile. Biological Report 85(7.5), U.S. Fish and Wildlife Service, Washington, D.C., USA.

Table 4. Sediment particle size distribution and Schoklitsch's sediment factor (s) for sediment samples collected on 15–18 November 1993 from Mugu Lagoon and its tributaries.

	Particle size distribution (%, dry weight basis)					
Site[a]	Pebble	Gravel	Coarse sand	Sand	Silt	s
01	0.0	0.4	1.6	92.5	5.4	0.023
02	0.0	1.5	2.5	94.9	1.1	0.026
03	4.8	8.2	4.5	81.4	1.1	0.070
04	0.0	0.2	0.2	93.4	6.2	0.022
05	0.0	0.0	0.0	92.1	7.9	0.021
06	0.0	0.0	0.0	88.7	11.3	0.021
07	0.0	0.1	0.1	89.6	10.2	0.021
08	0.2	0.1	0.1	88.0	11.7	0.023
09	0.6	1.4	0.2	87.2	10.6	0.029
10	0.0	0.0	0.0	82.2	17.8	0.019
11	0.0	0.0	0.1	98.4	1.6	0.023
12	0.0	0.1	0.2	99.2	0.5	0.023
13	3.7	1.3	0.1	88.3	6.7	0.047
14	0.0	0.0	0.1	84.8	15.1	0.020

[a] See Fig. 1 for locations of sampling sites.

were topsmelt, California killifish, arrow goby, shiner perch, and western mosquitofish. These results closely agree with findings from earlier fish surveys conducted in 1962–1964 (MacGinitie and MacGinitie 1969), 1972 and 1973 (Baker[7] 1976), and 1977–1982 (Onuf et al. 1978, Onuf and Quammen 1983, Onuf[1] 1987).

MacGinitie and MacGinitie (1969) and Baker[7] (1976) reported that fish were least abundant in the western basin of Mugu Lagoon, possibly as a result of severely restricted tidal exchange caused by culverts installed as part of road construction. During my study, fish from the western basin adjacent to Laguna Road (Site 07) were among the least abundant when compared to other sites in the lagoon (Table 2). However, in the western basin adjacent to South L Avenue (Site 09), fish were more abundant than elsewhere in the lagoon, possibly because sampling usually occurred in a shallow pooled area during low tide when tidal creeks and mud flats lying upslope were almost completely drained of water (Table 2).

Onuf and Quammen (1983) noted that distribution and abundance of several fish species in the eastern basin of Mugu Lagoon were strongly influenced by flooding during major storms in 1978 and 1980, which deposited thick blankets of fine sediments over the previously sandy bottom. Species spending most of their time in the water column (e.g., topsmelt; shiner perch; and bay pipefish, *Syngnathus leptorhynchus*) were adversely affected to a greater extent than bottom-dwelling species (e.g., California halibut, diamond turbot) because of a reduction in the low-tide volume of the lagoon and destruction of eelgrass, *Zostera marina*, beds that originally

[7] Baker, R.O. 1976. An ecological study of parasitism in estuarine fishes. Ph.D. Dissertation, University of California, Santa Barbara, California, USA.

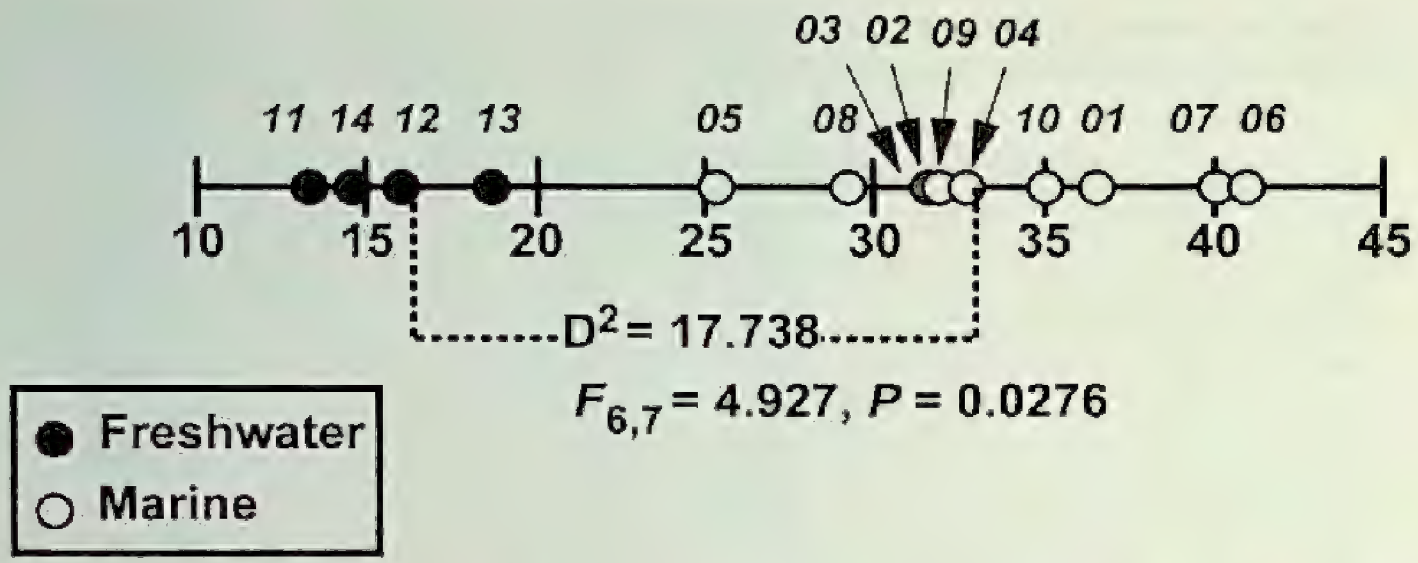

Figure 4. Simple discriminant analysis (SDA) for two groups of sampling sites (fish species assemblages) identified by cluster analysis. The SDA was conducted with ranked values of physicochemical data (water temperature, pH, dissolved oxygen concentration, salinity concentration, turbidity, and Schoklitsch's sediment factor). Four sites were correctly classified as "freshwater" and 10 sites as "marine" according to the minimum likelihood criterion of 24.84.

covered most of the eastern basin. During my study, no effort was made to measure the low tide volume of the eastern basin to determine if it had changed substantially since the study by Onuf and Quammen (1983). Casual observations indicated that eelgrass was absent from the eastern basin and elsewhere in the lagoon. As expected from Onuf and Quammen's (1983) findings, shiner perch were rarely captured in the lagoon, being abundant only in catches by bag seine in Oxnard Drain #2 (Site 10; see Table 2). In addition, no bay pipefish were captured during my study.

Compared to published information available for fishes in Mugu Lagoon, the status of fishes inhabiting Calleguas Creek, Revolon Slough, and the Oxnard drains is virtually unknown. Swift et al. (1989, 1993) reported that tidewater goby; steelhead, *Oncorhynchus mykiss*; and threespine stickleback, *Gasterosteus aculeatus*, historically occurred in Calleguas Creek. However, these three species have not been recorded from this vicinity for more than 2 decades (Swift et al. 1993). In 1975, western mosquitofish were captured from several reaches of Calleguas Creek and goldfish were collected in Revolon Slough (Wells and Diana[8] 1975). According to data reported by the California Toxic Substances Monitoring Program, brown bullhead, *Ameiurus nebulosus*, and fathead minnow, *Pimephales promelas*, have been collected from the Calleguas Creek watershed for analysis of environmental contaminants (Rasmussen[3] 1995 and earlier reports). During my study, seven estuarine fishes and five freshwater fishes were captured from tidally influenced reaches of Calleguas Creek, Revolon Slough, or the Oxnard drains (Table 2). Information summarized by Moyle (1976) and Swift et al. (1993) indicate that none of the freshwater fishes are endemic to the Calleguas Creek watershed.

[8] Wells, A.W. and J.S. Diana. 1975. Survey of the freshwater fishes and their habitats in the coastal drainages of southern California. Final report for Contract AB-26 submitted to the Inland Fisheries Branch, California Department of Fish and Game, Sacramento, California, USA.

Cluster analysis and association analysis performed with fish presence-absence data indicated that the 14 sampling sites could be classified into two groups, each populated by distinctly different fish species assemblages. One group (sites 01–10) contained marine species, whereas the other group (sites 11–14) contained mostly freshwater species. According to simple discriminant analysis of selected environmental variables, the two groups of sites were differentiated by salinity and, to a lesser extent, by temperature and dissolved oxygen. These findings support the generally accepted view that estuarine fish species assemblages are structured primarily by the tolerance of individual species to environmental variables such as salinity and temperature (e.g., Day et al. 1989, Moyle and Cech 1996). A host of physiological adaptations and behavioral mechanisms help estuarine fishes avoid extreme situations where their osmoregulatory or thermoregulatory abilities may be compromised (Day et al. 1989).

In addition to natural gradients of salinity, temperature, and other water quality variables created by the mixing of fresh and salt waters, degraded water quality resulting from pesticide-laden irrigation-return flows, nutrient-rich sewage discharge, or accidental spills of hazardous chemicals can affect the distribution and local abundance of fishes in Mugu Lagoon. According to T. Keeney (NAS, pers. comm.), fish die-offs occur occasionally in Oxnard drains #2 and #3. On 16 September 1993, a die-off involving striped mullet, western mosquitofish, and an unidentified freshwater crayfish (possibly *Procambarus clarki*) was observed in a 0.5-km-long reach of the northern branch of Oxnard Drain #3 (M.K. Saiki, unpubl. data). No other species were collected despite an intensive effort to capture fishes with the throw net. Several live mosquitofish were in distress, alternating between swimming erratically through the water in a "twirling" motion, then lying motionless at the surface. Measurements of water temperature (20.3–21.0°C), pH (7.7–8.4), dissolved oxygen concentration (9.8–25.7 mg/liter), salinity (2.7–3.7‰), and turbidity (12.4–19.3 NTUs) were within limits that the two fishes are known to tolerate (e.g., Moyle 1976). The only suspicious circumstance was the presence of numerous beige-colored floating particles, each measuring roughly 5–20 cm in diameter, that dissipated into an oily film when touched lightly with a stick.

Toxic pollutants can influence the structure of species assemblages in a variety of ways, including the elimination of all except the most tolerant species and the creation of conditions that favor species with opportunistic life histories (Sheehan 1984). One possible reason that the topsmelt is so ubiquitous in Mugu Lagoon may be related to biological characteristics that enable this species to rapidly invade marginal habitats during periods of favorable water quality (e.g., high abundance in most estuarine and coastal waters, extensive diurnal and seasonal movement patterns, omnivorous feeding behavior, and tolerance of a broad range of salinities and temperatures; Emmett et al.[9] 1991). On the other hand, occasionally lethal water

[9]Emmett, R.L., S.A. Hinton, S.L. Stone, and M.E. Monaco. 1991. Distribution and abundance of fishes and invertebrates in west coast estuaries. Volume II. Species life history summaries. ELMR Rep. No. 8, NOAA/NOS Strategic Environmental Assessments Division, Rockville, Maryland, USA.

quality conditions in Oxnard Drain #3 may be responsible for preventing the re-establishment of tidewater goby in Calleguas Creek and other brackish reaches of Mugu Lagoon. According to K. Lafferty (Department of Biological Sciences, University of California, Santa Barbara, pers. comm.), this fish occurs in the J Street Canal upstream from Ormond Beach and could enter the upper end of Oxnard Drain #3 by traversing about 3–4 km of lateral canals and other surface water connections.

ACKNOWLEDGMENTS

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NEW EQUIPMENT FOR PERFORMING MEASURED-DISTANCE DIVING SURVEYS

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A number of transecting methods are used in surveys to estimate the in situ abundance of fishes (Limbaugh 1955, Miller and Geibel 1973, Larson and DeMartini 1984, Bohnsack and Bannerot[4] 1986, Davis and Anderson 1989, Roberts 1995). While each transecting method offers practical or statistical advantages, it also has certain drawbacks. Permanent, pre-deployed transect lines effectively standardize transect length and are useful for making temporal comparisons at one site; however, they are often impractical for making spatial comparisons among many sites. Substantial effort is needed to construct and maintain permanent transect lines. Therefore, permanent lines are not cost effective for use at many sites.

Temporary transect lines can be deployed and retrieved in a single dive, standardize transect length, and are readily used to make multi-site comparisons. Time spent deploying and removing temporary transects, however, reduces the number of transects sampled per dive. In addition, changes in target species' behavior during temporary transect deployment often affect total numbers counted.

Transects using a set unit of swimming time and speed (timed-swim transects), rather than a set distance, are practical for making spatial comparisons among many sites (VenTresca et al.[5] 1996). Variations in swimming speed, however, may substantially change the unit of sampling effort. The resulting fish-per-unit-time counts are less useful than fish-per-unit-area counts in assessing populations.

We describe a rapid, measured-distance survey as a practical alternative to surveys conducted using permanent, temporary, or timed-swim transects. This method uses

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[4] Bohnsack, J.A. and S.P. Bannerot. 1986. A stationary visual census technique for quantitatively assessing community structures of coral reef fishes. National Oceanic and Atmospheric Association, National Marine Fisheries Service, South East Fisheries Center, Technical Report 41.

[5] VenTresca, D.A., J.M. Houk, M.P. Paddack, M.L. Gingras, N.C. Crane, and S.D. Short. 1996. Early life-history studies of nearshore rockfishes and lingcod off central California, 1987-92. California Department of Fish and Game, Marine Resources Division Administrative Report 96-4.

a retractable dog leash (Flexi USA Inc., 26 ft Standard model[6]) to lay out 10-m measured-distance transects. The dog leash is compact, inexpensive, easily operated, and most components are corrosion resistant.

Several modifications are necessary to allow use as an in-water transect line (Fig. 1). The leash cord is replaced with 2-mm flat-braided polypropylene transect line. The line is lightweight, thin enough to allow at least a 10-m length to fit on the reel, and does not absorb water. It also does not stretch, maintaining an accurate transect distance. The housing screws are replaced with stainless steel bolts and lock nuts. A small stainless steel washer is placed on the line inside the housing, 10 m from the free end, with a knot tied behind it. The washer creates a solid stop at 10 m and prevents over-winding of the reel spring. A brass clip is attached to the free end of the transect line. The clip is used to directly fasten the transect line to an object attached to the substrate (e.g., a small kelp stipe or sturdy coralline algae) or to a piece of "Romex" wire that may be wedged into a rock crevice. Holes are drilled in the housing to facilitate rinsing and lubrication.

The 10-m retractable transect lines were used by divers to make measured-distance surveys for spatial comparisons of benthic rockfish abundance among many sites. Each pair of divers used one retractable transect line, with each diver counting fish while swimming to the full length of the transect. Divers counted fish in an area 1 m on either side of the transect line and 2 m high (off the bottom), creating an observed volume of 40-m^3. Starting location and position within each reef (up-coast, center, or down-coast) were chosen at random from available reef areas. To eliminate depth bias, all transects were run perpendicular to the shoreline, across the reef area. On

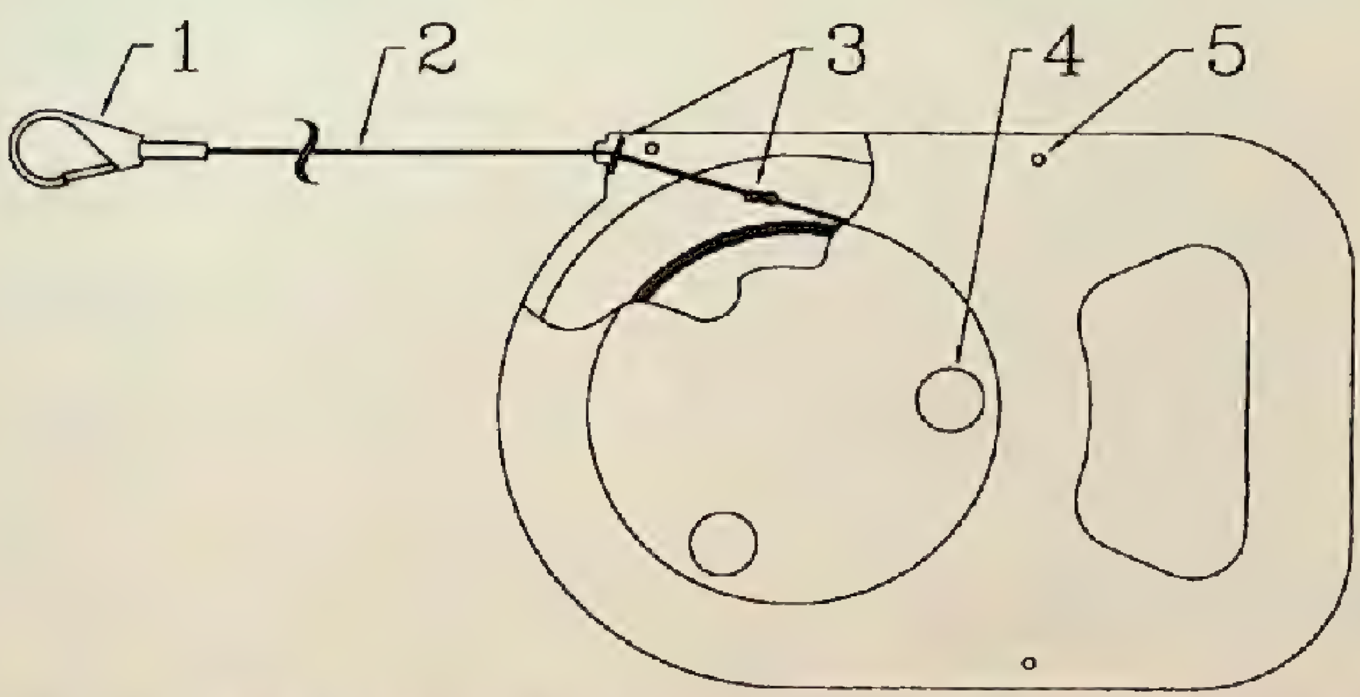

Figure 1: Retractable dog leash modified for use as a 10-m retractable transect line, showing (1) brass clip, (2) 2-mm flat-braided polypropylene line, (3) washer with stop knot, (4) drain holes, and (5) stainless steel bolts and lock nuts.

[6] Reference to trade names does not imply an endorsement by the California Department of Fish and Game.

each dive, five transects were surveyed in a line towards shore and five away from shore after moving to avoid overlap. After swimming to the end of a transect, a diver pulled sharply on the transect line to free the clip from the algae (or the Romex from a rock crevice), allowing the line to retract onto the reel. This method allowed multiple transects to be run in succession in one dive without pre-count disturbance. The surveys occurred during two California Department of Fish and Game Central California Marine Sport Fish Project research cruises. On the first cruise, nine divers surveyed fish along 252 transects. On the second cruise, 27 divers surveyed fish along 558 transects.

Research is ongoing to determine the relationship between estimates of benthic rockfish abundance from surveys using permanent transects and 10-m retractable transect lines. Future research is necessary to determine the value of retractable transect lines compared to temporary transects and timed-swim transects. Due to their ease of use and low cost, retractable transects would likely be useful in a variety of other diver-performed survey applications.

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